THE PAPARUA LANDFILL:
Hydrogeological, Geophysical and Hydrogeochemical Investigations of Groundwater Contamination by Leachate, Christchurch, New Zealand.

A thesis
submitted to fulfil
the requirements for the Degree

of

DOCTOR OF PHILOSOPHY

at the

UNIVERSITY OF CANTERBURY

by

VIVIENNE RUTH SMITH

UNIVERSITY OF CANTERBURY

1992

"Reality is ultimately water"

Thales (624 - 546BC)

ABSTRACT

The impact of leachate generation and migration from unlined municipal solid waste landfills on the Canterbury groundwater resource, was investigated at the largest site, the Paparua County Council Landfill. The 8 ha. landfill is located centrally within the western unconfined zone of the extensive 500 m thick glaciofluvial aquifer system. This unconfined area recharges deeper confined aquifers in the east, which underlie Christchurch City (pop. 300 000) providing its sole drinking-water supply. Between its conversion from a gravel pit in 1973 and closure in July 1978, the site periodically operated as a wet landfill and rates a high pollution potential classification (DRASTIC Index = 187, Le Grand +14G). However, leachate is confined to a 200 m x 200 m area in the northeast of the site, between the depths of 6 m and 9 m, by an underlying silty sand aquitard.

Landfill stratigraphy was determined from Wenner and Schlumberger soundings. Best-fit sounding models indicated leachate with a 2.3 to 6.7 Ωm layer, underlying unsaturated refuse (19.6 to 30.1 Ωm), and overlying silty sand (50 Ωm to 77 Ωm) and sandy gravels (>300 Ωm). Resistivity profiling at a=15 m and a=30 m spacings, revealed a low resistivity zone restricted both laterally and vertically to the northeast corner of the site. This zone, taken to represent leachate, exhibited slight southeast migration on the deeper penetrating profile. Down-hole nuclear geophysical logging did not detect the contaminant zone, but the landfill boundary was identified on a seismic traverse by a lack of refraction of the critical ray over refuse infilled land.

Beneath the 6 m deep landfill, a 1.5 to 8 m thick silty sand was detected in 4 lithologically logged wells installed at the site. The hydraulic conductivity of this unit at 2.9×10^{-9} to 3.2×10^{-7} m/s is significantly less than the 2.39×10^{-3} m/s of the underlying sediments. The silty sand therefore functions as an aquitard, producing a perched watertable in the overlying refuse and inhibiting downward leachate percolation as illustrated by resistivity pseudosections. In the deeper gravels and sandy gravels, point dilution tests showed that seepage velocities varied from 8.3×10^{-7} to 2.7×10^{-3} m/s and exhibited a strong relationship to uniformity coefficients of the sediments. Laterally continuous high velocity zones within deeper gravels could act as rapid dilution and transport zones for contaminants permeating through the aquitard.

Resistivity profiling dictated the installation positions of 16 monitoring wells. The presence of leachate was confirmed with groundwater samples from the northeast corner of the site contaminated with up to 160 mg/l NH_4-N, 0.11 mg/l Cu, 270 mg/l Cl, 210 mg/l Na, 2.8 mg/l B, high levels of Ca, Mg and high chemical oxygen demands. In these samples, total dissolved solids ranged from 1238 to 1736 mg/l and total organic carbon ranged from 58.8 to 98.7 mg/l. Amines, ketones and terpenes which result from the anaerobic degradation of putrescible refuse were detected. Alkyl benzenes, S- and N- heterocyclic compounds, and xylenes, at 5 mg/m^3 in one sample, were the only synthetic organic compounds found. In groundwater beneath the aquitard and downgradient of the northeast zone, these substances were below detection limits.

The landfill presently poses no significant threat to the quality of the underlying groundwater resource. However, leachate confinement by fortuitous hydrogeologic factors at the Paparua County Council Landfill cannot be taken as indicative of leachate behaviour at other unlined landfills in Canterbury, nor should the present site conditions be regarded as invariant. This study illustrates the importance of a site-specific investigation in gauging leachate effects, demonstrating the value of an integrated approach using techniques previously unapplied to New Zealand landfills.

TABLE OF CONTENTS

CHAPTER 1 INTRODUCTION

CHAPTER 1 INTRODUCTION

1.1 BACKGROUND

Awareness of groundwater pollution problems has heightened over the past few decades, as both our water supply demands and need to dispose of waste materials has increased. Not only are we realising that ongoing waste disposal must be undertaken in an organised and controlled way to avoid the deterioration of one of our most important resources, groundwater, but the spectre of past waste disposal practices returns to haunt us, necessitating a fresh look at the status and remediation of existing landfills.

In Canterbury, the North Canterbury Catchment Board (N.C.C.B.) originally carried out groundwater resource investigations under the Water Pollution Act of 1953 which related to the prevention of waterborne disease and protection of town water supplies. The Act emphasised increasing irrigation and potable water supply requirements, rather than quality control, and when the Act was breached it was only weakly enforceable. The superseding Water and Soil Conservation Act 1967 and its subsequent amendments (W.S.C. Act Amendment 1973 and others), was the first real attempt to look at water quality management. In 1978, the N.C.C.B. adopted Bylaw No.2 (Underground Water 1978) later replaced by Underground Water Bylaw 1990 where the primary function is the *"wise management and protection of the groundwater resource"* (Underground Water Bylaw 1990). The application of the Bylaw to the whole of the Canterbury region marked a new phase of groundwater management on the part of the N.C.C.B., typical of worldwide concern regarding such vulnerable resources. The responsibilities of the former N.C.C.B. were transferred to the Canterbury Regional Council (C.R.C.) on 1 November 1989.

Whilst previous research has examined the broad multilayered aquifer system in general terms, site-specific investigations had been undertaken only in response to isolated pollution incidents. The philosophy is changing to pre-empt such situations and focus on pollution prevention. Two factors lend an air of immediacy to this. Firstly the increasing concern about maintaining the existing high quality groundwater supplying drinking water to Christchurch City. Secondly it has been recognised that a number of specific sources, such as old municipal solid waste landfills, could potentially pollute the extensive permeable aquifer resource. Groundwater in the

unconfined area west of Christchurch City is the most vulnerable to contamination - in places heavily industrialised or used for intensive horticulture. Pollutants entering the aquifer system in this area can travel eastward into deeper confined aquifer zones, which public supply wells utilise for citywide water reticulation. In addition many residents in the western unconfined aquifer zone access their water supply from individual shallow (<30 m deep) household wells in the immediate vicinity of possible pollution sources. The C.R.C. initiated a multicomponent investigation into possible pollutant sources and their effects including: a regional survey of the location and condition of underground storage facilities and other pollution sources such as sumps, an investigation into herbicide/pesticide and chlorinated hydrocarbon concentrations in the groundwater, and an overview of waste disposal localities. Complementing the wider study, this investigation focuses on municipal solid waste disposal, specifically the former Paparua County Council Landfill - a site centrally located within the western unconfined zone, and its effect on the underlying aquifer system.

1.2 OBJECTIVES

During the summer of 1987/88, the project began with two primary objectives:

1. to evaluate the groundwater pollution potential from existing landfills in the western unconfined aquifer zone; and
2. to acquire hydrogeologic data, specifically dispersivity characteristics, from the shallow unconfined aquifers to assist predictive contaminant computer modelling.

Of particular concern is the area parallel to and east of Halswell Junction Road where a paleochannel of the Waimakariri River had downcut into the confining fine surface sediments. Here the unconfined permeable paleochannel extends eastward closer toward the central city. The groundwater also flows in this direction, recharging a large proportion of the confined aquifers beneath Christchurch City. Some 14 shingle pits have been extracted in this paleochannel in the past and many have been subsequently used for landfills. One site, that of the old Paparua County Council Landfill at Carrs Road, is studied in detail because:

1. it is the largest landfill site (both in area and volume) in the western unconfined aquifer zone;
2. it is close to a number of water receptors; single domestic wells and municipal supply pump stations; and
3. a number of incidents of water quality degradation have occurred in the

general locality.

Investigations were undertaken both at the Paparua County Council Landfill and at a second hydrogeologically similar site, which was considered to be free from leachate interference. This second location lies in a parallel Waimakariri River paleochannel, to the northeast on the Wigram RNZAF Airforce Base. At these sites the aim is to:

1. document the history of the Paparua County Council landfill;
2. establish the presence or absence of leachate at the site;
3. characterise the nature of that leachate, if present;
4. evaluate the potential groundwater quality degradation due to leachate at this location; and
5. characterise the hydrogeological behaviour of the gravel aquifers relevant to contamination transport.

This thesis includes a record of the development of the investigation to achieve these objectives incorporating the philosophy, methodology, and observed results from project inception to its conclusion.

1.3 METHODS

This problem in contaminant hydrogeology is approached dually as a scientific investigation and as an engineering project governed by the specific constraints of:

1. time;
2. money; and
3. available technology.

These are usual for groundwater investigations (U.S. Bureau of Water Invest. Manual 1985) though the variables are case dependent. Schwartz (1988) commented:

> *"I am often struck by the long time for expertise, knowledge and technology to diffuse into applications for solving practical problems. It is often said that experience is a good teacher, but not to the extent that it provides the rationale for making dumb mistakes using other peoples money"*

Ever mindful of this, the methods used in this investigation reflect the necessity to obtain scientifically valid data in a short time-frame using equipment at hand (specifically available or constructible in New Zealand) in as cost effective a manner as possible. Further restrictions include the applicability of hydrogeological investigation methods to gravel rather than sand aquifers, and the exclusion of radioactive techniques other than borehole logging. Invariably the focus is on

noninvasive geophysical techniques and point sampling.

A field based hydrogeological, geophysical and hydrochemical approach involving groundwater, leachate and aquifer characterisation and mapping is adopted. This broad but integrated method gives a "snapshot" of landfill behaviour and contaminant migration patterns. This is a holistic approach recognising the totality of the problem (within the given constraints) rather than fragments (Burges 1986). Increasingly, groundwater studies are becoming multidisciplinary, multicomponent, and multitechnological. Moore (1990) recognised this stating:

> *"consideration as a multitude of independent problems each separately solvable is an outdated concept".*

The long term behaviour of the site is considered, in that the investigation techniques used are simple and repeatable enough to be reused later, to monitor variations in the measured parameters with time. Likewise multipurpose boreholes are constructed with longer term use in mind. Thus this investigation is envisioned as the initiation of a monitoring programme whilst not fulfilling that purpose in itself due to time constraints.

The investigation follows a standard three phase investigation of solid waste landfills; an initial site assessment, an outline of the scope of work necessitated and a comprehensive site assessment, as outlined in Fabbri & Rodden (1990).

1.4 THESIS ORGANISATION

This thesis is presented in seven chapters. The remainder of Chapter 1 presents a summary of the regional climatic, geologic, hydrologic and social setting of the investigation. Chapter 2 introduces the concepts and importance of waste quantification and composition and presents the history of the Paparua County Council Landfill including its setting, site management, closure and final landscaping. In Chapter 3, the P.C.C. Landfill is examined at a feasibility level under a number of evaluation schemes to predict leachate generation and environmental impact. The results of a local reconnaissance groundwater quality survey are presented. Chapters 4 and 5 outline the geologic, hydrogeologic and geophysical investigations undertaken to characterise the site. The studies included the siting and installation of a number of monitoring wells from which hydrogeochemical samples are obtained. Both

inorganic and organic constituent analyses of in-situ and extractant leachate, and background water samples, are presented and discussed with consideration of the microbiological and chemical processes ongoing in landfills, in Chapter 6. An overview of the investigation including conclusions and recommendations are presented in Chapter 7. Where possible, investigation events are maintained in the chronological order of their occurrence to present the philosophy and controls influencing each stage of the investigation.

1.5 FUNDING AND SUPPORT

Financial support for this project was given by the University Grants Committee, the Paparua, and Waimairi County Councils (all of these councils are now amalgamated into the Christchurch City Council) and the North Canterbury Catchment and Regional Water Board (superseded by the Canterbury Regional Council). Technical and moral support and supervision was provided by the C.R.C., whose expertise was tapped to undertake pump tests and install point sampling monitoring wells. The Department of Scientific and Industrial Research (D.S.I.R.) Geology and Geophysics Division Groundwater Group were contracted to undertake downhole geophysical logging, supply equipment, and instruct in the use of electrical resistivity techniques. D.S.I.R. also provided access to the computer system and programmes to interpret geophysical data. D.S.I.R. Chemistry Division Water Section undertook all water chemistry analyses at Ilam, Christchurch (inorganic) or Gracefield, Lower Hutt (organic), except for those few undertaken by the the Cawthron Institute, Nelson. All laboratories are TELARC registered. Total organic carbon (TOC) analyses were carried out by the author using the analyser housed in the Public Health Engineering Laboratory, Civil Engineering Department, University of Canterbury.

McMillan Water Wells of Southbridge contracted to undertake all cable tool and air rotary percussion drilling. Waikato University dated wood samples by C_{14} analysis. Wood identification was carried out by Dr. B. Butterfield, Plant & Microbial Sciences Department, University of Canterbury using scanning electron microscopy.

1.6 REGIONAL SETTING

1.6.1 LOCATION

The Canterbury Plains on the east coast of New Zealand's South Island are a distinct

physiographic entity covering some 8000 km^2 and extending in a broad flat lying band from the Waipara River in the north to the city of Timaru to the south, and eastward from the Southern Alps and foothills to Banks Peninsula (Figures 1.1 and 1.2). Christchurch City the largest and most populous city area in the South Island (pop. 300,000) acts as an industrial, commercial and agricultural centre for the region. The city occupies an area flanking the northern Port Hills area of Banks Peninsula. The study locations of the Papaprua County Council Landfill on Carrs Road, and the Wigram RNZAF Air Force Base lie within the former Paparua County boundary to the west of Christchurch in the Wigram / Halswell suburban areas (Figure 1.3).

1.6.2 GEOLOGY

Basement rocks in the Canterbury region consist of extensively deformed indurated Mesozoic greywacke and argillite of the Torlesse Supergroup. The basement is unconformably overlain by Late Cretaceous and Tertiary marine and estuarine sedimentary rocks which are faulted and folded. Colluvium and alluvium material derived from structural highs infills depressions and is overcapped with younger Quaternary deposits consisting of coalescing glacial outwash and alluvial fans built up by the Waimakariri, Rakaia and other major Canterbury rivers. These fans emerge from the Southern Alps and foothills and have constructed the Canterbury Plains during the past 1 million years to connect the mainland to the "island" of Banks Peninsula, built up by volcanic activity between 5.8 and 11 million years ago. Alluvial fans dip gently seaward.

Much of Christchurch rests on the southern flanks of the Waimakariri fan. The surface of this fan slopes at 6 m/km inland at Halkett, and at 2.5 m/km near Wigram (Thorpe & Scott 1991). Glacial material, which is almost exclusively greywacke and argillite detritus, is found on the higher western parts of the fan while inter- and postglacial material, reworked by alluvial processes, is found eastward on the fan and abuts the eroded remnant Lyttelton volcano and the present coastline. Entrenchment of rivers has created both degradational and aggradational terraces, and a continuous supply of sediment from the mountains provide sediment for progradation. Rising inter- and postglacial sea levels resulted in periodic transgressions of the sea, with swamp, estuary, lagoon and beach deposits accumulating over the alluvial fans. The most seaward incursion during the Holocene was approximately 6000 years ago (Figure 1.4). Interaction of the progradational and transgressive processes has created a broad

complex gravel aquifer system up to 500 m thick (Wilson 1985, Figure 1.5).

Throughout the Canterbury Plains the indistinct aquifer layering is due to fine-grained glacial aquitards / aquicludes interbedded with coarser resorted and fresh alluvial material as aquifers. Westward lithologies coarsen resulting in an increased hydraulic conductivity. Within 5 to 15 km west of Banks Peninsula gravels constituting the uppermost aquifer (Riccarton Gravels) become confined and separated by marine shelly sands, silts, peats and clay of the Christchurch Formation which thicken and fine coastward (Figure 1.6). This surface confining unit interfingers with, and is overtopped by, gravel filled channels comprising the Springston Formation, which formed by changes to the course and rivermouth position of the Waimakariri River to the east. The Riccarton Gravel is thought to emerge from this cover approximately 40 km from the coastline at an elevation of 50 to 100 m below sea level (Talbot *et al.* 1986 after Herzer 1981).

Surface Quaternary stratigraphy has been detailed by Brown & Wilson (1988) following the work by Suggate (1958, 1963, 1965, 1973) and Brown (1973) on the basis of fan deposits and terraces, soil character, presence or absence of dune sand, degree of gravel and matrix weathering, and radiocarbon dating of wood and peat. Subsurface correlation from well logs is difficult and inconclusive where fine sediment may be missing, or where there are no shells or wood to date. In addition, erosion periods within gravels are unrecognisable, and gravels themselves include outwash and reworked material (Brown & Wilson 1988). Such a correlation however is presented by Talbot *et al.* (1986) on the basis of several hundred well logs and forms the basis of regional cross sections.

1.6.3 SOILS

Four distinct soil groups are outlined within the Christchurch locality (Raeside & Rennie 1974). These relate to four distinct landscape forms:

1. Port Hills Soils: developed directly on volcanic (predominantly basalt) rocks or on mantling loess deposited during Pleistocene glaciations;
2. Sand Dune Soils: bordering the Heathcote Estuary and paralleling the coastline northward;
3. Lowland and Drained Swampy Soils: underlying approximately 70% of Christchurch City (Talbot *et al.* 1986) with (originally) permanent or

periodic high water tables. These soils formed on the fine aquifer confining sediment of loess, flood alluvium and estuarine deposits; and

4. Waimakariri Fan Soils: formed on stony, relatively unweathered, dominantly greywacke alluvium of the Yaldhurst member of the Springston Formation - sometimes with a thin loess veneer, this group dominates the landscape west of Christchurch.

Soils within Paparua County are included in the Waimakariri group (Figure 1.7) wherein four physiographic units corresponding to soil age are recognised (Cox 1978) (Table 1.1). In the immediate vicinity of the P.C.C. Landfill, two of these physiographic units dominate: the low terrace soils (Waimakariri 700 to 2400 years B.P.) and floodplain soils (Selwyn <300 years B.P.). The units range from freely to excessively draining silt loams to very shallow, very stony soils, often overcapping previous soils. The paucity of runoff means that drainage patterns in the area are often indistinct.

1.6.4 HYDROLOGY

Principle aquifer recharge sources are shown in Figure 1.8. The Christchurch aquifers are river recharged, although rainfall and the contribution from unlined stock watering races becomes significant to the upper aquifers inland between the major rivercourses (Thorpe 1990). Oxygen isotope investigations have revealed that whilst groundwater recharge is generally derived from high altitude derived rivers, adjacent bores can reveal significantly different recharge contributions from precipitation (Taylor *et al.* 1989). Small quantities of recharge to surface gravels and the Heathcote River immediately adjacent to Banks Peninsula is derived from rainfall.

Pressure distribution, tritium, and chemical data indicates upward migration from deeper to shallower aquifers beneath Christchurch City via gaps in the aquitards. It is thought that the daily water abstraction, circa 300,000 m^3, via several thousand domestic, industrial and agricultural wells, may assist the upward flow. Evidence of downward leakage is found in the unconfined area to the west where surface derived contaminants (e.g. nitrates) have percolated to depths of up to 120 m (Talbot *et al.* 1986).

The groundwater quality in general is extremely high, both because the greywacke gravels constituting the aquifers are relatively inert to chemical dissolution (c.f.

Matthess 1982), and because flow rates are high, up to 250 m/day. Nitrate-nitrogen is found at elevated levels in the shallow aquifers south and west of Christchurch City attesting to pastoral agricultural land use. Iron and manganese are also found at elevated levels in some Canterbury areas (Talbot *et al.* 1986). Seawater intrusion is not problematic at present, as aquifer pressures are above mean sea level. The seawater / freshwater interface lies some kilometres offshore with groundwater discharging through the Riccarton Gravel.

Throughout the Canterbury aquifers, flow is from the west to the southeast, with hydraulic gradients averaging 3×10^{-4} m/m to 4×10^{-4} m/m for the deeper aquifers (2 to 4). In the confined zone beneath Christchurch City the shallowest aquifer has a hydraulic gradient of 3×10^{-4} m/m. To the west where the aquifer is unconfined, the hydraulic gradient is steeper at 1.5×10^{-3} m/m. In this aquifer, groundwater flow is from the northwest to the southeast, paralleling paleochannels of the Waimakariri River. At the northwestern corner of Banks Peninsula groundwater flow diverts eastward and southward around the volcanic obstruction to recharge the Halswell and Heathcote Rivers (Figure 1.9). The headwaters of these and other Christchurch rivers and their respective tributaries are found to the west of the City, where groundwater in the upper aquifer flows both under and over the surface confining unit. Flow patterns above the fine sediments is presumed to be dominated by the geometry of permeable buried river channels. With declining water levels, stream headwaters migrate downgradient and springs cease to flow. Springs known to flow in the Wigram / Halswell area were not witnessed during the period of this investigation.

Hydraulic conductivities vary from 10^{-4} to 10^{-1} m/s in the aquifers, with individual wells having specific conductance up to 60 l/s/m drawdown. Waimakariri River-derived shallow groundwater is calculated by Talbot *et al.* (1986) to take less than one year to reach the City Boundary from the Waimakariri River recharge seepage near McLeans Island, but an additional 30 years to reach the coastline due to the decreasing gradient and aquifer permeability.

1.6.5 CLIMATE

The Canterbury Plains constitute a relatively dry climatic zone as compared to the rest of New Zealand (Figure 1.10) with a mean rainfall occurrence rate of less than 6 events per month, with each event averaging <10mm, but both rainfall and its

distribution are highly variable year to year (Cox 1978). Received rainfall (approx. 1400 mm/yr) decreases eastward from the foothills of the Southern Alps, with 500 to 650 mm precipitation over the flat lying coastal region, and up to 2000 mm/yr over the more elevated parts of Banks Peninsula giving rise to some climatic diversity for those parts of Christchurch City built on the volcano's flanks (Figure 1.11).

Details for the three meteorological sites referred to in this text, Christchurch Airport, Wigram Airport, and Lincoln, are found in tables outlining monthly summed observations in Appendix 1.1. The sites are located on the Plains and are the closest sites with regular observations to the study locations. For these stations, rainfall is reasonably evenly distributed throughout the year, with an average monthly September/October low of 42 to 45 mm and a 64 to 75 mm May to July high. Rain is carried onto the Canterbury Plains by south to southwest winds, or by cool easterlies and northeasterlies originating out in the Pacific Ocean. Total annual pan evaporation is 1271 to 1329 mm and serious seasonal moisture deficits frequently occur during the summer months. During October to January gusty northwest föhn winds dominate the weather patterns exacerbating drought conditions across the exposed plains. The eastern Carrs Road, Wigram, locality of Paparua County is dominated by northeasterlies and easterlies (Figure 1.12), often while concurrent northwest föhn winds blow in the south and west of the County (Cox 1978).

Mean daily air temperatures range from a winter low of 5 to 6°C to a summer high (January / February) of 16.5 to 17°C, with mean ground temperatures from -1 to -2°, to 8 to 10°C respectively. Midsummer day length is approximately 15.5 hours; in mid winter the length is 9 hours. There are on average 2045 annual hours of bright sunshine. Frosts are common from May until September (late autumn to early spring), with an average of 86 days with frost (1949 to 1970 records), for the three meteorological stations nearest the study sites.

1.6.6 LAND USE / VEGETATION

Pre-polynesian Canterbury (<1000 AD) was forest mantled mixed podocarp and Kanuka dominated tracts (Cox 1978). Repeated burnoffs, both natural and human induced, altered the landscape by the time of European settlement (circa 1850) to tussock grassland and swamp vegetation. Initially settlers in Christchurch City obtained water supplies from the Avon River and from shallow wells dug into surface

gravel deposits. Drainage of the raupo swampland in the 1870's and the discovery of artesian water in the 1860's led to the settlement's rapid growth with agricultural expansion outward on the Canterbury Plains.

Present Plains land use is mostly intensive pastoral sheep farming although with more water available from the ground resource, agriculture and horticulture crops have become more commonplace on some soil types. High temperatures and long summer daylight hours mean that for optimal growth, plants require twice as much water as is available from average rainfall, hence irrigation is constant. In the northern Christchurch City hinterland orchard, berry fruit and some dairying is undertaken. To the west the better soils are sown in arable crops with other areas used for rural part-time farmlets and horse training establishments.

Land zoning (as at 1978) for the former Paparua County is shown in Figure 1.13; commercial, industrial and government land dominated the central area. The Paparua County Council Landfill, site 58 of the quarry locations, lies immediately south of a proposed but not completed motorway extension. The site is zoned rural general, a rehabilitated quarry site with pasture cover, in part occupied by a greyhound track with the remainder used for sheep grazing. The land forms part of the Municipal Electricity Department's Islington - Bromley 220 kV transmission corridor with two transmission pylons sited on it. The P.C.C. Landfill block is bounded to the west by an orchard and plant nursery block, to the south and east by pastoral land and to the north by an operational hardfill tip (Owaka Road 57a) and quarry (57b). Scattered housing (Figure 1.14) surrounds the site with a major housing development spreading northwestward from Halswell.

The Wigram Airforce Base - Government Defence Block lies to the south of the intensive industrial and commercial Sockburn zone with its associated housing. Industries sited at Sockburn include glassware and fertiliser manufacturers, timber yard construction and storage, bulk storage warehouses and an abattoir. The Base itself contains a housing area and operations centre in the northwest which includes aircraft hangers, workshops and control tower. The bulk of the land is used for grass landings, takeoffs, and flight clearance and security space surrounding the asphalt runways.

CHAPTER 2 WASTE DISPOSAL

CHAPTER 2 WASTE DISPOSAL

2.1 INTRODUCTION

Stapledon (1979) defined geology (or "earth-logic") as:

"finding out what is down there by understanding how it got there".

Adopting this approach it is necessary to examine the practices of municipal solid waste characterisation, waste disposal by landfilling, and the specific disposal history of the Paparua County Council Landfill at Carrs Road, to understand the leachate forming environment and consequently its effect on groundwater quality.

This report is limited to municipal as opposed to industrial or hazardous and special waste, as primarily municipal waste was deposited into the Paparua County Council Landfill. The categories are not mutually exclusive however and some evidence exists that other waste types were accepted into the site (discussed in section 2.4.3).

Municipal waste is predominantly composed of domestic household and garden waste collected by local authorities or privately disposed. It also includes waste from catering, shops, offices and light commercial and manufactured wastes entering the municipal waste stream (Buchan *et al.* 1991).

Hazardous waste is used as a broad term to denote industrial byproducts and waste materials discarded from homes, commercial establishments, and institutions. To be hazardous it must pose an unreasonable risk to human health, safety, property and the environment (Corbitt 1989). Such waste may be reactive, flammable, corrosive, toxic, pathogenic, carcinogenic, bioconcentrative, or persistent in nature (MWD 1983).

2.2 PAST WASTE MANAGEMENT

The three basic means of solid waste disposal (neglecting recycling) are: composting, incineration and disposal in landfills (Matthess 1982). Of these, landfilling is the most widely employed means of disposing of municipal waste (Clark & Piskin 1977, Lema *et al.* 1988, Cheremisinoff *et al.* 1984, Moss 1988, Thorstensen 1990). The low cost in monetary terms may not necessarily be low cost in environmental terms. Although the impact of waste disposal is site specific, landfilling has been called the least desirable method of dealing with waste, although no waste disposal method is without limitations (U.N.E.P. 1988 in Boulay 1991). The practice of landfilling is universal throughout the

local authorities in New Zealand (Refutech 1986) and common throughout rural areas with most farms and some industries operating their own sites either as landfills or "dumps", certainly not as "sanitary landfills". There is a need to recognise this characterisation (USEPA 1973) since each disposal option is very different in terms of operation and environmental and social impact (Cheremisinoff *et al.* 1984). Gunn (1983) summarised the New Zealand setting:

> *"...most dumps (landfills) are poorly sited, adjacent to groundwater or subject to flooding or tidal immersion... there are always sites in New Zealand which pose potential hazards".*

Taylor (1991) in Buchan *et al.* (1991) goes so far as to state:

> *"... the general standard of landfill design, construction and operation in New Zealand is appalling".*

Cocks *et al.* (1989) states:

> *"In the good old days when landfills were dumps, or at best tips, site selection was not very difficult. You just, for example, simply filled in the hole left by the gravel pit, or chose a convenient site close to town which was not too expensive to develop. It was even better if the council owned the land anyway, and if it created a bit of a nuisance, then so what! Visiting the tip was a recreational activity, and recycling of useful materials took place at the tip face. Many a kids' bike, recycled washing machine or old banger of a car has been kept going through bits gathered this way. Many a shed has been filled up with junk bought back from the tip as emptied through visits to the tip. In those days no one cared too much about environmental impacts, and no one really knew what leachate was".*

In New Zealand, disposal to "sanitary landfills" is a recent phenomenon with designed sites restricted to the larger metropolitan areas or to troublesome hazardous waste sites as exemplified by the Waireka Chemical Dump site near New Plymouth (Bell & Wilson 1988). Many New Zealand landfills are uncontrolled and it is difficult to ascertain either the number of landfills in any area or in some cases their locality. Further to this the task of quantifying the amount and components of waste produced and subsequently landfilled is made awkward by a number of factors the most obvious of which is the historic attitude to landfilling. Municipal solid waste becomes "out of sight" and therefore "out of mind" to the householder and industry once it is collected from their door, and to the local authority once it is received into the landfill and buried. Thus few records of waste quantity or components are kept as a matter of course. The characterisation of municipal solid waste is complex but has been attempted, as outlined in the following sections.

2.3 WASTE QUANTIFICATION

2.3.1 METHODS

Municipal solid waste varies for any area depending upon the amount of commercial and industrial activity and individual waste generation habits, and has been shown to vary seasonally, regionally and daily (Robinson 1986, Corbitt 1989, Synergy 1985). Any attempt at waste quantification becomes biased by these factors but there have been standardised approaches taken. Actual waste quantification is often carried out, as outlined in a study of Wellington's landfills by Moss (1988), where volumes are assessed by estimating refuse truck carrying size and the frequency of visits of the vehicles to their respective landfills. Robinson (1986) outlines the requirements and procedure for undertaking such a standardised waste quantification and composition survey.

Waste volumes are often linearly related to the size of the population (Corbitt 1989). Where municipal solid waste is dumped privately or where no tally is kept the refuse is quantified in cubic metres of the site occupied. As a consequence of the approximate nature of these methods, the correlation between waste generation and population is often not very good.

2.3.2 NEW ZEALAND / INTERNATIONAL EXAMPLES

Estimates of New Zealand municipal solid waste generation for each of the provinces during 1982 are given in Table 2.1 and are derived from a study undertaken by Synergy (1985). It serves to illustrate that the Canterbury district with Christchurch as the main centre, whilst representing 10.7% of the population of New Zealand produced 12.7% of the country's total waste. The higher domestic waste rate is somewhat offset by the lesser rate of commercial and industrial waste generation for Canterbury compared to the New Zealand average as outlined in Buchan *et al.* (1991) after Sherriff (1988), and shown in Table 2.2. Given the problematic nature of the quantification outlined above and the unregulated nature of refuse disposal these figures serve only to suggest that Cantabrians produce more than their fair share of rubbish. Moreover at first inspection, these figures would seem to dispel the environmentally aware image of New Zealand. Dividing the total annual waste by the population to achieve a "composite unit waste factor" (Robinson 1986), reveals that New Zealanders averaged a waste production of 1.75 kg/person/day (3.86 lb/person/day) based on 1982 figures. Estimates for the United States (Franklin 1988)

for the same year show lesser waste production at 3.04 lb/person/day. On the same basis Zanoni (1973), estimated 2.406 kg/person/day for the United States but that rose to 4.54 kg/person/day when all commercial, industrial and household waste is included. Hence assigned values are highly variable.

Table 2.3 lists some previously reported per capita waste production estimates for New Zealand and selected other countries. The wide range of these values is probably a function of the difficulty of obtaining a representative sample for analysis, and indicates the need to treat these figures with caution. The comparison between N.Z. and U.S. figures mentioned earlier may be less meaningful. Given the range of estimates for a single year from different sources (e.g., Sheriff 1988, where waste estimates from the Canterbury Regional Council and the Canterbury City Council vary by 12%), such quantities for evaluating waste production change over time, or for future predictions of waste generation are best used as a guideline only.

Temporal total municipal solid waste production estimates for both New Zealand and the United States are presented in Figures 2.1 and 2.2. Whilst the total solid waste production has steadily increased with time for both countries, current trends indicate the landfilled component to be levelling off in the USA, with more materials recovered for reuse, or consumed for energy - a process presently not used in New Zealand. Direct comparison between the two countries (Figure 2.3) shows New Zealand to have relatively stable annual municipal solid waste growth, whilst the US is more erratic. According to the Department of Health, New Zealand currently produces more refuse per capita (1.9 kg/day) than many other industrialised nations based on values of solid waste collected by local authorities (" The Press", 25 June 1990, in Boulay 1991). Globally the World Resources Institute (1990) (in Buchan *et al.* 1991), currently rates New Zealand about fifth in terms of municipal solid waste production per capita. However, these figures do not examine the types and concentrations of waste produced which affects the environmental impact of its production. Waste constituents are examined in Section 2.4.

2.3.3 PAPARUA COUNTY COUNCIL LANDFILL

Records of volume or weight totals for refuse deposited at the Paparua County Council Landfill are sparse. Waste amounts for each of the Christchurch Metropolitan Tips for 1976/77 is given in Lamb & Forbes (1977). The P.C.C. Landfill handled

60,000m^3 of uncompacted refuse during this period, of which 65% (39,000 m^3) was collected by the local authority; the remainder was delivered by contractors and private disposal. A volumetric calculation of total waste accumulated at the P.C.C. Landfill (Appendix 2.1), indicates approximately 5.8 x 10^5 m^3 of uncompacted refuse material, slightly less than multiplying the earlier local authority 1976/77 figure over the number of years of the sites use.

2.4 WASTE COMPOSITION

2.4.1 METHODS

Just as municipal solid waste quantification is inexact, so is the evaluation of its composition, due to its *"inherent heterogeneous and poorly defined aggregation properties"* (Franklin & Assoc. 1988). A wide range of material types, densities, and moisture content is possible. Surveys of municipal solid waste composition are carried out for a number of reasons, the focus of which dictates the assessment procedure. There are two basic approaches:

1. SITE SPECIFIC SAMPLE ASSESSMENT.

Samples of fresh waste are collected by truckload or are split-sampled (Corbitt 1989), and are then either visually inspected or physically sorted and weighed. A standard method for assessment is outlined in Robinson (1986).

2. MATERIAL FLOWS APPROACH.

Waste is calculated by tracing resources from primary production into the waste stream, using product statistical data (such as that listed in yearbooks) where:

Primary production + Imports + Exports + Tonnage Recycled - Tonnage in use = TONNAGE IN THE WASTE STREAM.

This method is used in Synergy (1985) (included in Table 2.4) and in Franklin (1988) to give municipal solid waste quantities and composition. A comparison of amounts for each constituent type gives a relative composition of total wastage.

There are limitations to both methods. Visual inspection alone is dependent on the skill and experience of the person undertaking the assessment. Sample sorting is subject to wide seasonal, regional and daily fluctuations in content, and there is disagreement as to the minimum weight of sample necessary to assess waste composition. Recommendations range from 136 kg to 3 tonnes per sample (Synergy

1985), up to one truckload for each sampling run, and with thorough documentation of the collection route carried out several times per year (Robinson 1986). In addition the minimum number of samples to accurately characterise municipal solid waste from a population of 3.14 million (approximately the population of New Zealand) has been quoted from an Australian study to be 9000 households (Synergy 1985).

Alternatively the materials flow approach provides only a broad indication of composition since some factors such as product decomposition are not quantified and estimates of "in use" resources either over, or under, estimate total tonnages entering the waste stream (Synergy 1985). Despite these error sources, Franklin Associates Inc. (1988) have used the materials flow approach to quantify municipal waste production since 1960 for the United States (see Section 2.3.2).

2.4.2 NEW ZEALAND / INTERNATIONAL EXAMPLES

Municipal solid waste composition based on sample sorting has been assessed in a number of studies, the results of which are compiled in Table 2.4. Constituent ranges vary greatly between the studies. Paper (including cardboard) constitutes 22 to 59.2% of the refuse weight in all studies except privately deposited refuse reported in Synergy (1985) where no paper was found. Other constituents include; inert material (glass, plastic, ceramic, rock) 19 to 50.5%, and putrescible (organic) matter including wood 23 to 57%. New Zealand studies (Douglas 1977, Synergy 1985, King *et al.* 1989) produce results similar in variability to U.S. data, but Douglas (1977) notes a factor that complicates a direct comparison between the two countries:

> *"in overseas local authority areas direct public access to the tips is prohibited. The 'weekend' outing of Christchurch householders to take their garden rubbish, lawn clippings etc. to the tip is therefore unknown... Christchurch is the 'garden city' and in this alone its refuse disposal problem becomes unique".*

In this Douglas refers to plans to change tip dumping over to a sanitary landfilling scheme at Bottle Lake (Section 2.5.3.2). A decade later, (1987/1988) that municipal landfill, serving an estimated population of 291,434 people from the Christchurch Region, received waste from the sources shown in Figure 2.4. Public disposal, constituting 36% of landfill received waste, is around 1.7 times the amount collected by local authorities. Other researchers put the ratio higher. Cocks *et al.* (1989) quote:

> *"generally householders carry 2 to 3 times more waste to the landfill than is carried by Council collection service",*

and Lamb & Forbes (1977):

"about half of the total refuse disposed of at the tip is carted in by private firms in trucks... or by members of the public in cars and trucks".

Consequently whilst constituent quantification is unavailable, putrescible matter is considered a significant component. The point is also illustrated in the Synergy (1985) study (Table 2.4) where putrescible food and garden waste accounts for 18% of the total waste stream, but a visual assessment of privately dumped waste showed it to be 47.2% by weight and 57% of household waste. King *et al.* (1989) presents data for the Auckland region, listing putrescible waste as accounting for 45.2% of refuse composition, whilst Douglas (1977) reported 13%. No mention in the last case is given of sampling size or method.

"As delivered" municipal solid waste can have a moisture content as high as 95% (U.S.E.P.A. 1973), leading to rapid biological degradation and leachate production. Most of the moisture comes from food and yard waste (lawn and tree trimmings etc.) as shown by Corbitt (1989) who summarised the moisture distribution of 27673 lbs of raw municipal refuse over a 5 day sampling period. Food waste contained 83.6% moisture, yard waste 71.6%, and total refuse moisture was 47.4% which was reduced to 28% when assessed minus the yard waste component. Thus, the high content of organic matter in New Zealand landfills, originating from homeowner/gardener access to disposal sites, will influence both the rate of refuse degradation and subsequent leachate production. Consequently the quantity of leachate, and more importantly, the type of leachate constituents, produced from municipal solid waste landfills in New Zealand, may differ from similar settings investigated overseas.

2.4.3 PAPARUA COUNTY COUNCIL LANDFILL

There are no records of waste qualification for the Paparua County Council Landfill and policing of toxic/hazardous content was incomplete, but the disposal facility is assumed to be typical of New Zealand municipal landfills both in content and disposal method. Information given in later municipal solid waste quantifying studies is taken as the best indication of refuse content (Table 2.4).

Freeze & Cherry (1979) state that 90% of industrial waste that is considered hazardous, is disposed of in landfills because of cost. Not surprisingly, Boulay (1989) revealed that in the Christchurch industrial suburbs of Opawa, Ferrymead, Woolston West and South, over 99% of all hazardous waste produced from small quantity

generators is disposed of by landfilling. This is due to ease of disposal, low cost, lack of information about proper disposal, and reluctance or unwillingness to change disposal practice. There has never been a specific facility in Christchurch to treat known toxic, hazardous, or liquid/sludge waste from industrial and chemical processing establishments, and farms (Bowden & Ayrey 1981). Such waste, often amassed by industrial waste collectors, was placed in the P.C.C. Landfill. On occasion there were (unconfirmed) reports of septic tankers offloading at the facility.

Cocks *et al.* (1989) cite a preliminary waste survey in Central Otago where

> *"horticulturalists generally consider the method of disposal of empty pesticide containers and waste pesticides as satisfactory. These methods include incineration and dumping, either on a farm or at a nearby public tip".*

The same social environment appears to have existed during the 1970's at the time of the Paparua County Council Landfill operation, an area surrounded by horticulture. Further to this, little incentive existed for the collection or recycling of waste oil or batteries, although prior to 1986 a tax existed on virgin oil to encourage recycling. Seventy million litres of lubricating oils are used annually in New Zealand; some is recycled, but almost half is unaccounted for (Little 1990 in Buchan *et al.* 1991, Fraser 1990). Most used oils are probably poured down drains, deposited in landfills or left to soak into the ground. Cocks *et al.* (1989) describe waste oil as:

> *"quantitatively the most significant hazardous waste generated".*

Additional materials of concern are as outlined in Boulay (1991) who cites a "Press" (8 Feb. 1984) article:

> *"surveys carried out on landfills in New Zealand have revealed that Council landfills contain solvents and thinners, paint and paint residues, herbicide and pesticide residues, arsenic compounds, heavy metals (e.g., lead and mercury) and many other chemicals, many of them poisonous, carcinogenic and mutagenic".*

The Paparua County Council Landfill's contents are therefore best described as undifferentiated household and industrial refuse, mixed with commercial waste including factory food processing and other unknown forms (Bowden & Ayrey 1981).

Though not constituting a representative sample of refuse composition, material collected from a borehole on the Carrs Road site gives some indication of waste materials found in the landfill (Chapter 4), as do photographs of part of the site (Figures 2.5 and 2.6).

2.5 LANDFILL CLASSIFICATION

The term "landfill" encompasses any site where waste is disposed of by land burial, and the term could conceivably be extended to cemeteries. Waste can include sludge, liquid, radioactive, toxic, hazardous and municipal solid refuse. With regard to landfill design and operation, several categories are notable.

2.5.1 SANITARY LANDFILLS

The American Society of Civil Engineers define a "sanitary landfill" as a refuse disposal facility wherein there is:

1. spreading of waste in thin layers,
2. compaction to the smallest possible volume, and
3. daily cover (U.S.E.P.A. 1973, Robinson 1986).

Alton Jameson (1989) extends the requirements for a state of the art sanitary landfill to include: lining of the site, leachate collection systems, gas control, sophisticated operating plan, site closure and landscaping, accompanied by comprehensive and continued monitoring procedures. The P.C.C. Landfill site includes none of the requirements for a sanitary landfill. It is not a designed site, but a "dump"; the need for recognition of this characterisation is pointed out in U.S.E.P.A. (1973). The long term decomposition of waste, its interaction with infiltrating water, and the subsequent formation of landfill leachate was not considered.

2.5.2 LEACHATE MIGRATION

With respect to leachate migration two types of landfill exist:

1. containment sites, and
2. natural attenuation or "dilute and disperse" sites.

1. CONTAINMENT SITES

Landfills are classified as containment sites where leachate is minimised or prevented from infiltrating the underlying soil/rock strata or groundwater. This is commonly achieved through the use of impermeable layers, including base liners, impermeable surface caps (usually clay),and more recently including flexible plastic membrane liners in "dry tomb" landfills (Lee & Jones 1991). For containment sites, where leachate leakage is detected, the contaminant zone can be isolated using a cutoff wall, where captured leachate is removed by pumping. Such "retrofit" landfills are costly, not only in design, but because large quantities of leachate are produced that must be

transported from the site.

A variant of the containment landfill is a "zone of saturation" site, where waste is placed below the water table but surrounded by impermeable clays (commonly >16 m thick, McGinley & Kmet 1984). Water levels are kept lower in the refuse than the surrounding site, so that flow is into the site, wherein collector wells abstract leachate for treatment.

2. DILUTE AND DISPERSE SITES

Dilute and disperse sites are those where leachate discharge is unhindered, and where natural attenuation reduces leachate to acceptable environmental levels within acceptable distances from the site (Whittle & Swanson 1986). This occurs through the processes of adsorption, exchange, precipitation and biological decomposition, and through groundwater dilution and dispersion.

The P.C.C. sites' behaviour is that of a "Decay A" type, dilute and disperse landfill (Knight 1984a after Kenley 1979), involving unaided leachate discharge to the natural underlying environment.

2.6 HISTORY OF THE PAPARUA COUNTY COUNCIL LANDFILL

2.6.1 SITE ACQUISITION

Originally the Paparua County Council shingle pit bounded by Carrs and McTeigues Road's (Figure 2.7), was excavated to supply gravel and sand for motorway works. Aerial photographs show this practice as early as 1961 where the old course of the Henui Stream is seen clearly to wind through and partly define the site. The pit was excavated to an approximate depth of 4.5 m below the level of Carrs Road (Watson *pers. comm.* 1989), into and just below the water table. It was last quarried in 1973 (Paparua County 1982).

When quarrying ceased, the 20 acre (8.09 hectare) site seemed an ideal location for disposal of household refuse, and typifies the commonplace New Zealand tip. It was "ideal" in terms of being "a convenient hole in the ground" in which to tip waste, typical of the practice of many developed countries in the past (Alton Jameson 1989), and "convenient" because of its proximity to the population base (Paparua County 1982, Figure 1.14). The P.C.C. site was first established as a municipal disposal location circa

1972/73, when all the local Christchurch metropolitan area councils (except Riccarton Borough), operated their own tips (Lamb & Forbes 1977). At that time Paparua County Council planning regulations did not require a conditional use hearing (which involves public submissions and debate) for the approval of the scheme.

Christchurch City, including Paparua County, then produced 200 tons of household rubbish per week ("The Press" 16/10/73). Lamb & Forbes (1977) indicate that 60 000 m^3 of waste was accepted annually at the site, of which 40% was annual Local Authority disposal, and the remainder industrial and household disposal. The Carrs Road site became the main Paparua County municipal refuse disposal location serving a population base derived mainly from the Halswell, Sockburn and Hornby areas as shown in Figure 1.14, but it also received waste from Riccarton Borough.

The total population in the area steadily increased, with Paparua County numbers swelling by 15.3%, from 27,147 persons to 31,313 (Table 2.5). The bulk of this population growth occurred in Halswell, accounting for one third of the increase, between 1971 and 1976. This trend is still observed, with farmland northwest of Halswell, but southeast of Carrs Road the area is being redeveloped as suburban housing.

A further consequence of population rise was the possibility of extending the southbound four lane highway to cope with traffic flows to and beyond the Wigram area from Central Christchurch (Figure 1.13). Investigations in the late 1960's and early 1970's of the proposed route along Wigram Road and across the top of McTeigues Road, incorporated a corridor of relatively undisturbed ground to facilitate road construction. This extended several metres to either side of the existing McTeigues Road, and formed a nonextractable, northern boundary to the Paparua landfill. Several shallow foundation test wells were driven along the route, the significance of which becomes clear in relation to water levels in the area, as discussed in Section 3.2.3. The roadway project was put on indefinite hold, and there are no immediate plans for its construction.

2.6.2 SITE OPERATION

Weekly collections of household refuse, from bins and of (newly established) paper bags ("The Press" 7/6/72), was the main contributor to the tip, but during its operation

trailer loads of garden material (lawn clippings, shrubbery) and hardfill were also common. Members of the public paid a toll fee to drive on to the site, and offload refuse over the pit edge. This practice is common in New Zealand, and is clearly shown in 1975 aerial photographs of the site. Some felt the payment costly and, typical of poor waste management sense (Warren 1989), subjected the roadways to "fly-tipping" or illegal dumping along the road verges. This still occurs along the roads bounding the P.C.C. Landfill. Plastic bags were not in general use and were not commonly observed at the site during its operation (Noonan *pers. comm.* 1990), although several feature in Figure 2.6. The dumping of animal matter was permissible, if arrangements were made with the tip staff to have the material covered ("The Press" 18/8/77). A single plant operator compacted refuse, on site with a light bulldozer (Watson *pers. comm.* 1989). Waste was sometimes burnt off, watered to keep dust down, or was lightly covered with gravelly soil in an operative manner loosely resembling the "progressive slope method" (Robinson 1986), but no systematic covering system existed.

2.6.3 SOCIAL / ENVIRONMENTAL EFFECTS

Site operation practices as outlined earlier gave rise to a "lively" fly population, wind blown refuse deposition on surrounding paddocks, dust, stench, fire risk ("The Press" 17/1/78), and rat infestations. Locals attest that rats were as big as cats and liked to forage through horse feed in nearby farm buildings when the pickings at the dump became leaner. These side effects are typical of open rubbish dumping practice (Alton Jameson 1989, Warren 1989, Corbitt 1989). With its adverse environmental impact, the operating site was a contentious one for local residents, who objected to both its siting and site management. There were continual complaints, and a petition lodged about its operation to the Paparua County Council.

Following pit excavation, the hollows into which refuse had been placed were subject to rising groundwater levels. Concern was periodically expressed about this, with the recognition of possible contamination of the local potable groundwater supply ("The Press" 18/8/77).

At times, the rising water table due to increased rainfall began to pose operational problems ("The Press" 29/3/77 and 18/8/77). At one stage the County was depositing refuse into water, but attempts were made to separate the disposal area from the water by a shingle bar (Lamb & Forbes 1977). During this period, however, water rose in

places to < 1 m from normal ground level at McTeigues Road, into the deposited refuse (Bowden & Ayrey 1981). During a two to three week period in August 1977, several trucks of offal and two sheep carcasses were illegally dumped after operating hours, into a pond on the eastern, Carrs Road, side of the tip ("The Press" 18/8/77). Figures 2.5 and 2.6 record this episode with sheep carcasses and other refuse floating in water ponds. The local newspaper quoted the Paparua County Council Engineer:

> *"About two years ago rubbish was being dumped into ponds at the tip. Although it was not polluting water supplies, we considered it undesirable to dump rubbish into water... the latest carcasses and offal would not affect water supplies."*

The carcasses were immediately covered over with soil. Locals have never been at ease with the dump's siting, and the same newspaper article speculated that the carcass dumping episode may have been done to deliberately embarrass the council and force closure of the site. There were no reports of groundwater-sourced water supplies becoming contaminated in the immediate vicinity following the dumping incident, although there is evidence of ground water quality degradation in the general vicinity in later years (Bowden & Ayrey 1981).

2.6.4 SITE CLOSURE

In the mid 1970's investigations were ongoing into the establishment of a sanitary municipal waste landfill, north of Christchurch City. Leachate taken from an open pond at the P.C.C. Landfill (S84:522 923) was sampled for its chemical constituents, during the high water level period in 1977. Leachate from the P.C.C. Landfill was applied to a test area in the Waimairi Coastal Strip as part of the investigation into the establishment of the Waimairi Landfill at Bottle Lake, north of Christchurch City, and the associated Metropolitan Refuse Transfer and Recycling Stations (Noonan *et al.* 1977, Rouse 1977). This regional scheme was designed to replace the existing dump sites around Christchurch, such as the problematic P.C.C. Landfill, with three Metropolitan Refuse Transfer Station collection points, and transportation of refuse to a lined sanitary landfill north of the city. One notable feature of the Bottle Lake scheme was the encouragement of more active recycling of waste. Following assessments of the feasibility of compost manufacture in Christchurch (Mulcock & Johnson 1971), experimental recycling and composting of material at the P.C.C. Landfill was tried. This was an attempt to solve some of the environmental problems, but served only to accelerate them. The Health Department recommended the site close *"as soon as an alternative site could be found"* ("The Press" 28/2/78).

Investigations into alternative sites (pending the outcome of the Waimairi facility) had been underway, but the proposed and favoured venue at Pounds Road, near Christchurch Airport, did not gain approval ("The Press" 29/3/77, 25/8/77).

The provisional decision by the Paparua County Council to join the Bottle Lake Metropolitan Refuse Disposal Scheme ("The Press" 31/1/78), made in early 1978, was confirmed shortly afterward ("The Press" 3/2/78). The P.C.C. Landfill subsequently closed on 31 July 1978 (Watson *pers. comm.* 1989). Refuse from the area was transported to the old Waimairi County Council Landfill on Johns Road (Bowden & Ayrey 1981), until the opening of the sanitary landfill site at Bottle Lake in July 1984.

2.6.5 SITE FINISHING

During its operation, the topsoil put aside to cover the P.C.C. Landfill, had been used up. With the expectation that the Pound Road alternate venue would become available, the topsoil had been used in 1977, to all but seal over the site. Lamb & Forbes (1977), summarising local sites quote with reference to Carrs Road:

> *"this tip's life is virtually ended, unless the material is mounded above the general road level, there is no more than a few months of use remaining. Proposals for other locations and interim disposal at the Waimairi tip are being investigated".*

When the new venue proved unavailable, further refuse accepted at the P.C.C. locality became mounded over the almost completed site above road level. Local residents suggest this overcapping and layer build up occurred a number of times during the life of the tip, but this cannot be confirmed. The result, however, was that the closed tip received minimal imported surface cover, and the "finished" site had refuse clearly protruding through the soil, with patchy uneven soil cover.

Leased later, with conditional site amelioration terms involving the spreading of further truckloads of topsoil, the site has slowly been converted over to sheep pastureland, and recreational space, including a Greyhound Track. Present site topography (Figure 2.8), clearly delineates the tipped area as being 2 to 4 m higher than the surrounding pastureland, with refuse in places protruding through the poorly developed soil cover. Its visual landscape impact is low, as recorded in a regional survey of old quarries in the area, two years after tip closure (Paparua County 1982). Therein it is classified as quarried and rehabilitated land, under a rural general zoning, but the report does not record its past use as a major landfill facility.

Whilst leases apply to recreation groups and a local farmer, the land ownership was retained by Paparua County Council. In November 1989 this local body amalgamated with the other County offices into the Christchurch City Council, and the site is now overseen by the Property unit of that organisation.

2.7 PRESENT WASTE MANAGEMENT

In 1978 all tip operators were advised by the North Canterbury Catchment Board to cease dumping of wastes, other than inorganic hardfill, into tips below a level likely to be affected by groundwater, and to cease disposal of all organic and chemical waste likely to affect groundwater quality. The establishment of other refuse disposal facilities was opposed under the Town and Country Planning Act. In 1979 Paparua County Council amended the Operative District Scheme relating to the filling of excavations and depressions, to fulfil the same requirements (Bowden & Ayrey 1981). Several violations of these regulations, in other tips along the western quarry belt, have since been successfully prosecuted through the court system. These follow violations of the Act but are only a retroactive approach; the full protection of the groundwater resource is still governed by individual decisions as to disposal practices.

Environmental problems have not been solved with the new Waimairi site, which currently accepts 600 tonnes of waste per day. Although classed as a sanitary landfill (Robinson 1986), the site is unlined, and without a methane collection system. Revegetation on the first 11.4 ha. completed stage has proved unsuccessful (Buchan *et al.* 1991), and the continued threat to shallow water table groundwater is suspected. Lee & Jones (1991) discuss the possibility that sanitary landfills, and even "dry tomb" landfills, merely postpone water quality problems until their respective containment mediums fail, thereby relegating the concerns to future generations. This idea is promoted in Buchan *et al.* (1991), and continues to be the focus of present day waste management concerns.

2.8 SUMMARY

The Paparua County Council Landfill which was originally a gravel quarry, became a landfill facility for undifferentiated household and industrial refuse in the early 1970's. The estimated 5.8 x 10^5 m^3 of waste deposited at this site was never thoroughly quantified or qualified. The municipal solid waste landfill is assumed to be typical of

New Zealand dumps and is expected to contain some hazardous and sludge waste including heavy metals, arsenic compounds, herbicides, pesticides, halogenated hydrocarbons and other undefined poisons, and carcinogenic or mutagenic substances. A significant proportion of waste was putrescible, with approximately half deposited by household "weekend" conveyors rather than Local Authority collection. Due to high organic matter content, leachate may significantly differ from that found in overseas examples.

Site operation and maintenance at the unlined, undesigned site, was minimal. Refuse was lightly bulldozer compacted and burnt off, or covered when required. As a "Decay A", dilute and disperse type landfill, operation practices induced detrimental social and environmental effects. These included insect pests, vermin, litter and potential leachate pollution of the groundwater supply. Leachate effects were exacerbated with refuse directly dumped into groundwater, or water levels rising into previously dry deposited waste. When officially closed on 31/10/78, the site was improperly finished, and only subsequent remedial work has enabled it to be used as pastureland and recreational space. At present the site has a low visual impact, but the subsurface and long term effects of its existence were unknown prior to this study.

CHAPTER 3 LEACHATE GENERATION

CHAPTER 3 LEACHATE GENERATION

3.1 GENERAL CONCEPTS

A landfill constitutes a potential pollution source localised in space and time and is therefore classed as a point source (Moore 1990), from which the greatest environmental problem posed is leachate production (Clark & Piskin 1977, Cheremisinoff *et al.* 1984, Whittle & Swanson 1986, Lema *et al.* 1988, Warren 1989).

3.1.1 TERMINOLOGY

Leachate has been variously defined, but is generally considered to be a liquid, or combination of liquids, produced by the percolation of water through the solid waste in a landfill (Cheremisinoff *et al.* 1984, Todd 1980, Freeze & Cherry 1979, Fetter 1988, Fenn *et al.* 1975). The water takes up the organic and inorganic substances (including suspended matter, pathogens and dissolved chemicals) from physical extraction and from hydrolytic and fermentative processes ongoing in the landfill (Clark & Piskin 1977, Lema *et al.* 1988). In the absence of a basal liner, the resultant leachate is transported into the underlying hydrogeologic environment by downward percolation through the vadose zone. If groundwater is already in contact with waste, leachate will immediately develop below the water table.

Leachate can mix with and contaminate groundwater, introducing organic and inorganic constituents above the background levels found prior to leachate development. Inherent then in the assessment of groundwater degradation is the necessity to measure and define the background values of groundwater constituents (Matthess 1982).

Characteristically the leachate forms a plume that usually spreads in the direction of groundwater flow, and decreases in concentration with increasing distance from the source. Such plumes can be complex, however, in that some leachate constituents are aqueous phase liquids (APL), whilst others are nonaqueous (NAPL), and can either float at the water surface (light NAPL) or sink deep into the aquifer (dense NAPL). Aqueous and nonaqueous liquids can interact as solutes/solvents for each other, as emulsions, or can be colloidally suspended (Knight 1984b). Plume behaviour is therefore governed by complex effects including density and viscosity, and not purely by the groundwater flow. In addition, the ongoing processes of solubilisation,

volatilisation, and adsorption, as well as the local effects of geology (layering, lenses and pore size) interact so that any one plume can have multiple lobes, occupying different aquifer zones, and consisting of completely different substances, thus complicating the investigation of such a setting.

Any contaminant plume is of concern given the potential pollution of groundwater. When safe human consumption or processes using water are restricted or prevented, and the fluid constitutes a hazard to human health, then it is polluted (Fried 1975, Todd 1988, Milde & Mollweide 1970 in Matthess 1982, Buchan & Key 1956). Matthess (1972) uses the term "anthropogenically polluted groundwater" to distinguish sites where pollution is a result of human - aquifer interaction, such as landfilling, and is not (as is possible) naturally induced. Polluted groundwater is assessed on the basis of the constituents that exceed acceptable health, palatability or aesthetic standards for water use (particularly potability) (Dept. of Health 1984). The standards differ according to both the water use e.g., crop irrigation versus drinking water, and the type of organism involved e.g., human versus fish, because the effects on each will vary.

The concepts of pollution and contamination must also be viewed spatially and temporally because of the importance of where, and at what time, water is sampled. With regard to dilute and disperse landfills the concept of compliance zones have recently arisen. In Illinois for example, pending regulations require contaminants issuing from a waste cell to attenuate to applicable groundwater standards within the lesser of 30.5 m, or the distance to the property boundary. The compliance distance is applicable for a 100 year period from the time of waste placement (Hensel *et al.* 1991).

Compliance zones may be unnecessary in some cases because of the attenuation of contaminants in the unsaturated zone, however, this is largely governed by the buffering capacity of the underlying sediments (Mather 1989). In aquifers where neither time nor distance exists for contaminant attenuation between refuse and water, compliance zones are still applicable.

3.1.2 EXTENT OF THE PROBLEM

Given that millions of landfills exist worldwide, relatively few had been found to be the cause of potable groundwater pollution up to the 1980's (Freeze & Cherry 1979,

Ham 1975, Zanoni 1971). However, more recent researchers project a more alarming picture. For example the United States Environmental Protection Agency estimates that of approximately 75,000 landfills in the United States more than 75% of them pollute groundwater with leachate. Lee & Jones (1991), note that of California's 2,200 landfills, about 300 have been investigated and 80% of these were found to be polluting groundwater. What is not stated is that most of those investigated would have been researched because of the underlying aquifers high susceptibility to pollution, the importance of the aquifer, or some existent indication that contamination is occurring. Thus these landfills are probably those most likely to pollute. In addition, the scale of the pollution and the full impact on the aquifers is not explained. Are some of these sites only polluted metres away from waste deposition because of soil adsorption mechanisms? Are the pollutants of low mobility, or does microbial action break them down rapidly? There are too few details given. Thus while the extent of leachate problems appears overwhelming, it may be that site specific factors tend to mitigate many pollution events. This is notwithstanding the perspective of Lee & Jones (1991) that slow stabilising landfills are merely problems whose time is yet to come - the "timebomb" case.

The problem is conceptualised in Figure 3.1 as resulting from three primary factors: leachate generation, leachate composition and leachate migration, all of which are discussed in greater detail in later sections. Only in those landfills where leachate is generated is there the possibility of groundwater contamination. Generation may be immediate (where waste is tipped directly into groundwater), may develop with time, or it may never occur. The process is not reversible in that groundwater could, for example, reconcentrate back in the refuse, but it is "two-way", in that leachate production can stop, and a site originally contaminating an area can become stable and non-leachate producing. Secondly, leachate pollution potential (hazard) depends on the leachate composition which is a function of the original waste constituents (section 2.4.2) and their concentrations. Finally, human, animal and environmental health or aesthetic endangerment will occur as a function of leachate migration where contact with a pollutant is made over a period of time, for which there is a calculable risk of exposure, dependent on the exposure time, and the inherent hazard. These factors must be judged in perspective with the comments of Lee & Jones (1991) however, as they relate only to the constraints operating presently. Future site conditions around landfills will change and typically more people will live closer to older sites, and the

likelihood of failure of protective liners around waste increases with time; but the questions raised about pollution scale and real extent are still relevant. The popular notion that all landfills pollute and endanger the environment (Stone 1991, Boulay 1991) is therefore a misconception.

Municipal solid waste landfills (MSWL) fare somewhat better in their environmental impact than landfills generally. From a 25% sample of the 6000 MSWL sites in the USA, the USEPA estimates that while only 15% are designed with liners and 5% have leachate collection systems, only 10% of the total are in violation of some state groundwater protection standard (Hickman 1990). Farvolden & Hughes (1976) in Freeze & Cherry (1979) found that any solid waste can be buried in almost any site if it is properly designed and operated. The fact is, nevertheless, that "proper" design and management is a relatively new phenomenon, and the longer term effects of liner failure in engineered sites are only now being examined. Older landfills are of particular concern for reasons outlined by Gebhardt & Jankowski (1987). These are:

1. site planners were less likely to consider groundwater during their site selection processes;
2. sites were less likely to be designed, rather they can be considered "dumps";
3. practices were less concerned with hazardous waste, perhaps allowing indiscriminate disposal of toxic waste; and
4. landfill users were less informed about hazardous waste disposal.

As illustrated in Chapter 2 all of these notions are applicable to the Paparua County Council Landfill Site.

Although leachate can have a higher Biological and Chemical Oxygen Demand (BOD and COD) than urban sewage, and be more toxic than many industrial effluents (Price 1987, Lema *et al.* 1988), an understanding of the environment into which this potentially hazardous brew percolates is required in order to determine the environmental effects. Ham (1975) states:

> *"the concentration of a pollutant in leachate from a landfill is in itself of no real consequence in the surrounding environment ... if it is found only in the initial volumes of leachate issuing from a fill just before reaching field capacity".*

The concentration and volume of leachate produced, its dilution and transport in groundwater, and its interaction with the geological materials, must all be incorporated

in an evaluation of the leachate's real pollution significance. The systematic evaluation of all landfill sites either existing, or planned, is therefore necessary (Barber *et al.* 1981) to distinguish highly pollution potential sites for which further investigations toward landfill design or hydrogeologic remediation is required.

3.2 GENERATION EVALUATION METHODS

3.2.1 INTRODUCTION

Leachate generation is dependent on:

1. refuse conditions;
2. landfill ground surface conditions;
3. underlying sub-surface conditions; and
4. water availability (Lu *et al.* 1985).

Of these, the critical factor affecting leachate quality and quantity is the amount of water allowed, or able to flow through the solid waste (USEPA 1973, Fetter 1988). In general the higher the water transmittal rate through a landfill, the greater is the amount of leaching, but the more naturally dilute and attenuated the leachate becomes in the geological and groundwater regime beneath the landfill (Fenn *et al.* 1975). It is recognised that in some circumstances physical and chemical processes may be unable to cause any significant change to the toxic substances within a leachate plume (Freeze & Cherry 1979). Conversely in general, the less water transmitted, the higher is the leachate concentration. Absence of water means that leachate will not be generated unless the waste is already sodden when deposited. Through the landfill itself, leachate strength increases with depth.

Water transmittal is dependent on highly site specific climatic and hydrogeologic factors, hence leachate production, concentration, and migration are also site specific (USEPA 1973). Several simple evaluation methods are currently used to assess a landfill's leachate generation potential, one of which, the Water Balance Method, is examined and applied to the Paparua County Council Landfill. Should leachate generation be indicated or assumed for predictive reasons, other methods can be applied to assess the leachate impact on the surrounding environment. In this study the input data for predicting site susceptibility for leachate generation comes from readily available published sources (e.g., soil maps and meteorological reports). Each site can be rated at a feasibility stage, prior to any subsequent in-situ site investigations into the design or remediation of a landfill.

3.2.2 WATER BALANCE METHOD

The Water Balance method originally published by Thornthwaite & Mather (1957), with regard to moisture availability and transfer in soils, was republished and applied to solid waste disposal sites by the U.S. Environmental Protection Agency (Fenn *et al.* 1975) to predict leachate generation. Various workers have subsequently amended the method for computer use (Scharch 1985), to include improved meteorological data embracing subtler climatic factors (Holmes 1984), or larger (spatial and temporal) databases. Others have used enhanced numerical (Thompson & Tyler 1984) or analytical (Schoeder *et al.* 1984) computer models based on the Water Balance Method to predict leachate generation. The HELP model of Schoeder *et al.* (1984) which allows modelling of up to nine stratigraphic layers in a landfill representation may be of long term predictive use at the P.C.C. Landfill site if the inherent assumptions of the method are met. Unfortunately this approach requires considerable meteorological data, and a detailed understanding of the hydrogeological nature of the site, obtainable by in-situ analyses which are not appropriate at a feasibility stage of investigations.

In terms of a simple evaluation of leachate generation at the P.C.C. site, only the original approach of Fenn *et al.* (1975) is attempted. This models a landfill as a two phase system (Figure 3.2) where water availability to contact waste is a function of precipitation supply and the action of the soil cover, as would be the case at a properly sited "dry" sanitary landfill. However, other possible moisture sources to a landfill include: water added during the operation, fluid incursion through the sides from surrounding formations, initial waste moisture content, water from decomposition, and groundwater moving up into waste. In Fenn's approach these water sources are assumed negligible. Further assumption of the method are as follows:

1. the landfill is completed with 0.6 m of final cover and graded to 2 to 4° slope over most of the landfill surface;
2. computation applies after placement of the final cover;
3. the final site use is open pasture or recreation land;
4. a fully vegetated surface cover is of moderately deeply rooted grass drawing water from the soil only;
5. soil cover and compacted waste are uniform in terms of hydraulic characteristics in three dimensions; and
6. the landfill depth is much less than its horizontal extent, hence all water

movement is vertically downward.

The first condition would be satisfied for parts of the P.C.C. Landfill, but at others the cover is thinner, with refuse protruding through the surface. Similarly moisture sources may or may not be available to contact waste at the site. Nevertheless for the purposes of this feasibility exercise, **the P.C.C. Landfill is treated as a dry tip for which the water balance assumptions are assumed satisfied.** This is a simple approach but serves as a working model for the ongoing investigation.

In Fenn's approach, water percolation is via two phases as illustrated in Figure 3.2. Soil factors such as vegetation, soil type, and runoff, and the particular soil in contact with the atmosphere determines the amount of water available to percolate through waste in Phase 1 - the transition of fluid through the soil media. Thereafter, the waste is assumed to act as a soil in Phase 2, such that any channelled flow of leachate, due to the heterogeneous nature of the refuse, occurring prior to the waste reaching field capacity is assumed discontinuous and negligible. The refuse moisture storage capacity therefore determines the quantity, and time, of leachate first appearance (Fenn *et al.* 1975), since the waste layer must be bought to field capacity before the water surplus will percolate into the underlying strata.

The field capacity of solid waste has been determined to range from 20 to 30% by volume (Fenn *et al.* 1975, after Fungaroli 1971). Similarly a range of 10 to 20% by volume has been attributed to the solid waste moisture content, although as mentioned in section 2.4.2 other researchers have found values up to 90%. For this exercise, field capacity is taken to be 30% (300 mm of water per metre of refuse), with an initial moisture content of 15% (150 mm/m). Thus one metre of refuse will absorb the difference between these values (150 mm/m) before the field capacity is exceeded and leachate appears at the base of the refuse layer. Based on this solid waste moisture absorption value, Fenn *et al.* (1975) presented nomographs where given the landfill depth, surface area, and annual site percolation from the water balance calculation, both the expected time of leachate appearance and the quantity generated, could be estimated.

3.2.2.1 Paparua County Council Landfill

For the Paparua County Council Landfill, the finished site features a deep rooted pasture, corresponding to a crop runoff coefficient of 0.15, with a root zone of 0.6 m.

The original Waimakariri and Selwyn Formation soils around the P.C.C. Landfill are all variants on a silt or sandy loam with low available water capacities (< < 120 mm). Whilst the top cover of the landfill site is a reintroduced and establishing soil, its characteristics are assumed to be similar to the original soil types in the Carrs Road locality. Meteorological records for the three monitoring stations closest to Carrs Road (Lincoln, Christchurch Airport, and Wigram, (Appendix 1.1) vary only slightly, and for this exercise Christchurch Airport values are used. Monthly averaged meteorological data from Christchurch Airport for the period November 1953 to November 1984 (N.Z. Met. Serv. 1986) are used to produce Table 3.1. The data are calculated using the Thornthwaite method model of Coulter (1973), although as suggested in that publication, calculated potential evapotranspiration (E.T.) values are lower in the drier areas of New Zealand during dry periods than actually occurs. In Table 3.1, water balances for three nominal soils with available water capacities (A.W.C.) of 40 (a), 80 (b), and 120 mm (c) are presented, along with a worst case scenario (d).

For the P.C.C. Landfill, with an averaged waste depth of 5 m, the worst scenario would be to treat it as an uncovered waste pile for which no soil cover is provided, and no runoff occurs. In this (Phase 2 only) situation, 163 mm of percolating water is available yearly for downward movement. Leachate would first appear 4.2 years following the placement of the waste (Figure 3.3) at a rate of 14.23 x 10^6 litres per year. Spread over the 8.09 ha. site this is equivalent to an annual liquid layer 17.6 cm thick. The leachate generated would then be a moisture contribution to the geologic material underlying the landfill which functions as another filter phase (Phase 3). However for any soil cover with available water capacity and associated soil storage as low as 40 mm and 24 mm respectively, substantial direct downward movement would occur. However, on the basis of a monthly average water balance, the Canterbury climate is such that no water is available for percolation. This is an equivalent case to the Los Angeles example presented in Fenn *et al.* (1975) and suggests that no leachate would generate at the P.C.C. Landfill if it was a dry landfill with precipitation as the only source of water.

However, rainfall events vary in intensity and individual storms may produce sufficient water to significantly alter the moisture conditions of the landfill. If the mean maximum 1 day precipitation events are used and balanced against Priestly estimates

of mean daily potential evapotranspiration as in Table 3.1e, then individual 1 day rainfall events could contribute as much as 16.8 mm of percolation to the landfill. In the period 1945 to 1985 for which these mean 1 day rainfalls are calculated, maximal 1 day rainfall events have been as high as 110 mm of which 89 mm would have percolated downward (Table 3.1f). Obviously these are extreme values, however it serves to suggest that a monthly averaged balance is insufficient to predict the moisture behaviour at the site. In addition, the history of the P.C.C. site (Chapter 2) outlines periods in which waste was deposited directly into groundwater, while the Water Balance Method assumes this source of water to be negligible. As a consequence of both these points, predictions of leachate generation using the monthly averaged Water Balance Method is misleading and there is a need to assess the extent of groundwater in contact with refuse.

3.2.3 WATER LEVEL OBSERVATIONS

Confirmation of groundwater behaviour at the site comes from water level observations in well M36/0890, located approximately 100 m northeast of the Carrs Road and McTeigues Road intersection. The well was driven as a foundation test hole as part of the motorway extension investigations mentioned earlier. It is the shallowest well in the area, originally driven to a depth of 5m. Throughout the operational period of the P.C.C. dump and up until 1989, monthly water levels were recorded in this well. Thereafter, the bottom of the well became filled with road metal. Attempts made during this investigation to clear the blockage (by forcing water down into the well and surging the infilling material out), failed, but the well is still openly accessible to water entry. A water level probe will now only reach to a depth of 4.4 m. The top of the well casing at 19.51 m above sea level, also represents the level of the roadway adjacent to the landfill. The lithologic log shows the well to penetrate and end in continuous gravels. Thus the water level observations in M36/0890 are assumed to represent the behaviour of groundwater at the base of the P.C.C. dump, ignoring the possibility of groundwater mounding due to refuse.

Figure 3.4 superimposes details of the site history onto the water level record of M36/0890. The period 1967 to 1974 was relatively dry with depths of 4 to 4.5 m to the water table recorded. This correlates with the maximum gravel extraction depth indicated by Watson (*pers. comm.* 1989) and to gravel extraction practice, excavating material to, or just below the water table. Water levels rose considerably during the

operation of the site as the P.C.C. Landfill from 1974 onward. Site activity from various literature sources (Chapter 2) correlate well with recorded water levels. As shown during the P.C.C. Landfill operation two important processes clearly occurred:

1. refuse was deposited directly into water, and
2. groundwater rose up through in-situ previously dry dumped refuse.

Attempts such as pond infilling in early 1977 to prevent the first process was foiled by later water level rises. The "carcass dumping" episode occurred at a groundwater level peak. It is ironic that the date of the site closure coincides with a further peak, the highest level to which water levels rose throughout the area during the period of level recording. Groundwater levels since 1978 have never risen above 2.5 m below ground level (17.3 m a.m.s.l.) and have frequently subsided to below the original 5 m well depth. Since the latest peak in 1986, water levels have dropped to beneath 4.4 m below ground level making the levels unrecordable.

Water level observations from well M36/890 clearly indicate that groundwater not only contacted waste within the P.C.C. landfill but saturated it for extended periods during the site operation and in the years following. Leachate therefore immediately generated upon refuse deposition. The effect of subsequent falling water levels is unknown, but the possibility exists that leachate ceased to be generated. Regardless, leachate was at some stage produced. The effect of the leachate on the surrounding water quality, both laterally and with depth, and its movement away from the site is unknown. Certainly pollutants will be washed more rapidly, and in larger quantities, from a wet tip than from a dry tip, with biological and chemical stabilisation times lessened (Rouse 1977). To gauge the potential impact of leachate generated, two evaluation methods are examined to assess the requirement for site specific investigations.

3.3 IMPACT EVALUATION METHODS

3.3.1 DRASTIC

The DRASTIC system, designed as a regional evaluation and mapping scheme for areas of 100 acres of more (Aller *et al.* 1985), can be applied to smaller areas (Gebhardt & Jankowski 1987). Seven hydrogeologic rating factors, each multiplied by a weighting factor, are summed to produce a DRASTIC INDEX quantifying the environmental impact of leachate from a landfill. Usually indices range from the mid 20's with very low pollution potential to greater than 200 indicating high pollution

potential. The indices are grouped into variously coloured mapping units on a regional scale to allow an overall pictorial representation of suitable (cooler light blue shades) versus unsuitable (red for warning) planned landfill sites. Applied to existing sites the scheme gives an indication of those sites where preferential remediation programmes should be focused.

Table 3.2 illustrates the scheme applied to the P.C.C. site. A DRASTIC INDEX of 187 indicates that this area has a high pollution potential.

3.3.2 LE GRAND

Le Grand (1964) designed an empirical point count system based on the hydrogeological factors at specific landfill sites. Its application for Australasian use is examined in Kidd & Hancock (1983), who found the scheme to be a rapid and easy means of site assessment involving little time or expense. Originally based on landfill site physical features of; depth to the water table, sorption above the water table, permeability of the aquifer, water table gradient, and horizontal distance from the contaminant source to the water supply (Le Grand 1964), the scheme has been revised (Le Grand 1983) to include: thickness of the unconsolidated material over bedrock, and an assessment of the confidence in the assigned numerical ratings of each of the contributors. Further descriptors conveying additional site specific information have been added to the numerical rating.

Each site is judged in a series of steps, the first summing the sites' physical factors whilst the second stage evaluates a "degree of seriousness" from a nomograph relating aquifer composition and waste disposal practice. The P.C.C. Landfill site rates a "G" or high degree of seriousness, large in area or quantity and of moderate consequence. The degree of seriousness is subsequently compared with a "contaminant severity" in stage 3 where the nomograph point is transferred to a second isometrically divided nomograph of "Protection of aquifer ratings" (PAR). The line closest to the stage 2 point becomes a PAR, the lowest value that would adequately protect an aquifer of that hydrogeologic setting for that specified use. Subtracting the PAR from the site rating determines the "situation rating" which may be adjusted if there are engineering modifications such as the addition of a liner to the landfill. The P.C.C. landfill rates as shown in Table 3.3. The determined alpha-numeric descriptor (+14 G) represents a very probable contamination risk (situation rating > +8) with an almost certain

unacceptable status and the landfill is deemed an F site, the poorest possible rating.

3.4 LOCAL RECONNAISSANCE

3.4.1 INTRODUCTION

Water levels throughout the Canterbury Region are regularly monitored by the Canterbury Regional Council but do not provide specific local detail as to groundwater behaviour. The regional pattern of groundwater flow is northwest to southeast, with Paparua County straddling a hydrogeologic saddle (Figure 1.9) (Talbot *et al.* 1986). Flow directions diverge as groundwater flows around the flanks of the Banks Peninsula, discharging east in part as the Heathcote River and southwest in part as the Halswell River. Springs and artesian boreholes become prevalent southeast of Halswell in this area.

3.4.2 METHODS AND MATERIALS

A local reconnaissance survey, carried out in December 1987 in the vicinity of the P.C.C. Landfill, had the following objectives:

1. to outline a working hydrogeologic model of the area;
2. to delineate the local groundwater flow direction and gradient;
3. to quantify field measurable groundwater properties of pH, Eh, dissolved oxygen content, temperature and electrical conductivity; and
4. on the basis of 1 and 2 to delineate zones of degraded water quality.

Information gleaned from New Zealand Water Well Data Forms (held by the Canterbury Regional Council), revealed numerous wells potentially usable for groundwater parameter quantification. Wells were sufficiently spread throughout the area to ensure adequate, even, survey coverage.

Well logs derived from these records were collated to construct cross-sections of the local stratigraphy (Figure 3.5). Whilst in some localities fine sediments are recorded at the surface and/or within well logs, the lateral extent of these units is poorly defined. All save five wells included in the reconnaissance survey were screened at a depth of <36 m and are considered to penetrate into the uppermost aquifer which becomes confined to the southeast. The deeper wells; M36/3856 at 38.4 m (used for water level data only), M36/0740 at 95 m, M36/1212 at 109.1 m, M36/2566 at 86.8 m, and M36/3304 at 62.6 m, are included for comparative purposes of water levels and

field quality tests between these and shallower wells.

Site visits to one hundred and forty six wells revealed seventy two of them unusable for a variety of reasons including: well destruction since the last well data entry, unlocatable wells, blockages preventing water level probe access, wells temporarily out of action due to pump or casing maintenance, or the pumped groundwater was sampled and/or contaminated by the plumbing system from the well to the header tank. Seventy four wells were confirmed as measurable with use made as follows:

21 water levels only,

20 water level and field quality tests, and

33 field quality tests only.

The position and well record numbers of wells incorporated in the survey are given in Figure 3.6.

The survey was undertaken over a period of two days to ensure a nearly instantaneous snapshot of groundwater characteristics. Sampling for water quality from the nearest bleed point on each well is undertaken after a minimal 5 minute drawing time or after groundwater temperature stabilised. This ensures sample properties represent an influx of aquifer water rather than stagnant well water or the plumbing system. Portable electronic pH/Eh and conductivity electrode membrane meters record their respective parameters in bottled samples. Temperatures and dissolved oxygen (D.O.) levels are assessed using the handheld probe of the Y.S.I. (Yellow Springs Instrument) Model 58 Dissolved Oxygen meter held under running water. All equipment except the D.O. meter automatically corrected field values to the standard of 25°C. The possibility of cross-contamination is minimised by rinsing equipment with distilled water between samples. Well elevations were levelled in by surveying from permanent roading benchmarks located within the study area and water levels reduced to metres above mean sea level as listed in Appendix 3.1.

3.4.3 RESULTS

The reconnaissance survey results (listed in Appendix 3.1) are spatially presented in Figures 3.7 to 3.11.

3.4.3.1 Water Level Observations

Water level observations (Figure 3.7) indicate a very regular piezometric surface

dropping from an elevation above mean sea level of 16.0 m to 13.25 m across the area. This confirmed previous more coarsely spaced surveys indicating that the flow was from the northwest to southeast, parallel to the paleochannel orientation. The survey did not extend sufficiently south to detail the zone where flow diverges east and southwest along the foot of the Port Hills. A hydraulic gradient of 8.9 x 10^{-4} m/m is calculated for the surveyed area. Several wells south of Quaifes Road (notably the Fountains Road area) and near Sabys Road are artesian.

3.4.3.2 pH and Temperature

For the shallower wells, pH ranges from 6.1 to 7.9 with a very regular zone running between Halswell Junction and Awatea to Dunbars Road of 6.5 (Figure 3.8). A single well (M36/2447) exhibits an anomalous pH value of 9.1. As this reading was repeatable on correctly operating monitoring equipment, it is thought this bore may penetrate deeper than the well record indicates, as deeper wells exhibit a slightly higher pH range of 7.6 to 9.1. Groundwater temperatures range from 12.3°C to 15.4°C. Two wells, M36/2437 at a depth of 12.1 m, and M36/2805 at an unknown depth, provided water samples with temperatures of 16.2°C and 17.0°C respectively. These values are higher than is usual for Canterbury aquifers but have been known to result from water heating when a centrifugal pump is used for sample collection.

3.4.3.3 Electrical Conductivity

Electrical conductivity values range from 20 to 40 mS/m for the shallow wells, with an elevated conductivity of 100 mS/m from bore M36/1077, and lower values (down to 7.6 mS/m) along the southern end of Quaifes Road and north of Whincops Road (Figure 3.9). The five deeper wells display lower conductivity values of 8.5 to 14.0 mS/m. The conductivity range for all wells in the survey is characteristic of extremely high quality water (cf. Matthess 1982).

3.4.3.4 Dissolved Oxygen

Dissolved oxygen content recorded in mg/l is dependent on both atmospheric pressure (or equivalent altitude), and temperature. For the purposes of the survey, the first variable is neglected, as insignificant altitude changes are made between samples, and sampling is close enough to sea level to require no correction. Temperature variance however changes oxygen solubility. Maximal D.O. values for the temperatures of samples (12.4° to 17°) are 10.76 to 9.65 mg/l respectively. To standardise values for

sample comparison, recorded D.O. values are therefore converted to a % saturation of the highest possible value at the respective sample temperature. At several locations extreme oxygen content changes result as a function of erratic pump behaviour, with air periodically introduced into the sample. Whilst every care was taken to minimise aeration, it is recognised that oxygen enrichment by this means can occur without being aware of the process, and little control can be asserted over the various pump systems. Thus, recorded values are of guideline worth only.

Oxygen saturation ranges from 1.6% to 99% with a spatial distribution as illustrated on Figure 3.10. An oxygen reduced groundwater zone is located in a northwest to southeast zone central to the survey area, passing beneath the P.C.C. Landfill, although this appears coincidental. Values of oxygen saturation for the deeper wells are just as diverse (3.8% to 60.8%).

3.4.3.5 Eh

Reduction - oxidation potential ranges from -0.01 to -0.167 V with shallow wells in the central Awatea Road to Halswell Junction Road steady at between -0.100 and -0.03 V (Figure 3.11). Well M36/2447 exhibits a lower Eh of -0.167 V. Lower pH and Eh values occupy the same central NW to S central survey zone. As plotted on Figure 3.12, Eh - pH values lie along a regression relationship where Eh = 0.303 - 0.051 pH, paralleling the stability margins for natural waters. Deeper wells portray lower Eh values, from -0.069 to -0.166 V, but relate to pH in the same manner as found for shallow wells.

The results are taken as local groundwater quality background values. None of the general field testable quality parameters for either the shallow or deeper wells indicate groundwater degradation in the surveyed area. Water quality based on conductivities is extremely good (Matthess 1982) with no deleterious zones evident. No correlation is found for any of the field parameters with depth.

3.5 SUMMARY

Any landfill sites, particularly those under designed or mismanaged, have the potential to pollute groundwater, but not all landfills produce this effect. Leachate generation is a function of complex climatic, hydrogeologic and waste characteristics that may interact to actively enhance pollution, or alternatively, to preclude leachate generation

and migration. To evaluate a landfills pollution potential and pollution significance these factors must be considered collectively.

The P.C.C. Landfill site is an unusual one in that the feasibility evaluation methods traditionally used to assess the likelihood and consequences of leachate generation give conflicting results. The Water Balance Method using monthly averaged rainfall and evapotranspiration values indicates that under a Canterbury climate, no leachate would generate in a dry landfill where precipitation is the only source of water. However, the inadequacy of a monthly characterisation in order to predict leachate generation is recognised with the possibility of high intensity rainfall events. In addition, water level observations attest to groundwater contacting refuse so the landfill is, or has been, a wet site, but whether leachate is still actively produced is unknown. Both the DRASTIC and LE GRAND methods for evaluating leachate impact, indicate that the P.C.C. Landfill (or anywhere in the western unconfined zone of Christchurch City) is the poorest possible location for a landfill. It rates as an unacceptable risk with probable contamination/ pollution potential.

A reconnaissance survey of 74 shallow wells in the Paparua County Council Landfill vicinity, screened in the upper unconfined aquifer (<36 m), confirms the local hydraulic gradient (based on 41 wells), to be 8.9 x 10^{-4} m/m from northwest to southeast, toward the Port Hills. Field quality tests on 53 shallow wells indicate groundwater to have a temperature varying from 12.3° to 15.4°C (with two anomalous values of 16.2°C and 17.0°C), a pH range of 6.1 to 7.9, electrical conductivity values of 7.6 to 46.2 mS/m (with one anomalous reading of 119 mS/m), slightly reducing Eh range of -0.010 to -0.167 V, and dissolved oxygen saturation between 1.6 and 99%. The quality of five deeper wells (at 36 m, 62.6 m, 86.8 m, 95 m, 109.1 m) is similar but more restricted. Slightly greater pH values (7.62 to 9.1), lower Eh (-0.069 to -0.166 V) and conductivity (8.5 to 14.0 mS/m) ranges are found in these deeper wells. None of the parameters measured showed any correlation with depth. The values given are taken to constitute local "background" groundwater quality data and indicate that water quality is excellent. Eh versus pH values for all wells tested closely follow the relationship Eh= 0.303 - 0.051 pH, paralleling the normal stability fields for water. With the exception of D.O. results, which are considered unreliable due to pump behaviour, field testable quality parameters indicate no aberrant values in the vicinity of the P.C.C. Landfill. The pH values are lower than the New Zealand Water Quality

Guidelines desirable range, but this is usual for Canterbury aquifers, and other parameters indicate high quality water.

Whilst no water quality abnormalities are observed in the local reconnaissance survey, this is not sufficient to negate the existence of leachate contamination. Several scenarios may apply:

1) leachate is no longer generating and that which did in the past has been flushed through the system,
2) field testable quality parameters used in the survey are blind to the true constituents of the leachate,
3) the "dilute and disperse" nature of the landfill is such that leachate is sufficiently dilute as to have no deleterious effect on general groundwater quality indicators such as conductivity,
4) whilst leachate is generating, it is attenuated a short distance from the site, or
5) the leachate plume is existent and detectable, but is travelling in a zone or zones undetectable by existing wells.

An on-site specific investigation is required to assess each of these possibilities and establish the nature of leachate generation at the Paparua County Council Landfill.

CHAPTER 4 HYDROGEOLOGY

CHAPTER 4 HYDROGEOLOGY

It is assumed that the reader is fluent in general hydrogeology terminology including: groundwater, porosity, permeability, aquifer, aquiclude, aquitard, storativity, specific yield, transmissivity, hydraulic conductivity, advection, average linear (seepage) velocity, Darcy velocity, piezometric and potentiometric surface. Outlines of these concepts are given in any of the following texts: Todd (1980), Freeze & Cherry (1979), Bowen (1986), Price (1985), Fetter (1988), Bouwer (1978), Davis & DeWiest (1966), to which the reader is referred.

4.1 INTRODUCTION

Following the findings of the groundwater reconnaissance study, a site specific assessment involving the installation of boreholes was required at both the Wigram and Paparua County Council Landfill sites, to fulfil the following objectives;

1. to characterise the lithologic and hydrogeologic properties at each site; and
2. to undertake hydrochemical "source" monitoring at the P.C.C. Landfill and satisfactorily resolve answers for the absence of leachate from the reconnaissance survey.

To satisfy the first objective, a variety of methods were proposed. Wells were to be installed at both sites, and the well contents both lithologically and geophysically logged. In addition a tracer test was proposed for the Wigram site to assess typical groundwater hydraulic behaviour. At the P.C.C. Landfill, vertical access to intervals below the landfill base was required to assess likely leachate migration paths. Each well was thus required to be multi-functional, with the following operational constraints:

1. a minimum of 1 pumping well and 2 observation wells (within the zone of drawdown effect) at each locality for pump tests;
2. observation wells downgradient of an injection well for tracer tests;
3. lithologic logging was required of all wells with samples collected throughout the well depth in at least 1 well at each locality;
4. a fully slotted casing for wells subject to discrete interval tests or sampling;
5. available local technology restricted the well drilling method to air rotary or cable tool;

6. cost and material availability restricted the casing type to butt-welded steel sections, or bell and spigot glued PVC sections;
7. borehole geophysical logging at each locality required a minimum internal well diameter of 4 inches; and
8. successful geophysics and down-hole testing required minimal ground disturbance from well installation.

With well installation in unconsolidated sediments, a disturbed annulus is created about the casing which can have a marked effect on geophysical logging and on any methods used to characterise vertically-distributed aquifer properties. Installation methods assessed in Morin (1986), Morin *et al.* (1988), Keely & Boateng (1989) and Bennett *et al.* (1988) found the following increasing trend in borehole disturbance:

1. driven steel casing (least disturbance);
2. PVC inserted into a driven steel casing, followed by removal of the steel casing;
3. casing installed after mud rotary drilling; and
4. casing installed after auger drilling (most disturbance).

Because the first two techniques were used, borehole disturbance was minimised for the wells installed in this investigation.

Of the driven casing techniques, cable tool is usually the more expensive ($212/m) and slower, and there is the possibility of fluid contamination by water introduced above the water table to assist the functioning of a sand bailer sample collector. Whilst air rotary is faster and usually cheaper ($100/m), the possibility of groundwater contamination arises from the possible introduction of compressor lubricants in the introduced air flush system, whilst at waste sites the return air can bring volatiles to the surface contaminating the working site. It is important to minimise the introduction of foreign water and/or substances during the drilling process to avoid affecting later hydrochemical sampling.

Down-hole contamination is also possible by the sorption/desorption of materials from the well casing material. Although Riggs & Hatheway (1988) suggest that no *conclusive* evidence has been forthcoming on this topic, they suggest the use of steel casing where organics are suspected, and PVC where metal contamination is possible. Both methods and both screen materials were used in the installation of single screen

wells at the investigation sites. This conforms with the current trend away from multiple screened wells which suffer installation problems (Riggs & Hatheway (1988). Well construction methods are outlined in the following section.

4.2 WELL INSTALLATION

4.2.1 LOGGING WELLS

At each site, three wells were installed, the first of which was to act as a pumping well. This was a 150 mm (6.6 inch) internal diameter steel cased well (M36/6102) emplaced by the cable tool technique to a depth of 36 m at Wigram, and to 30 m at the P.C.C Landfill site (M36/4017). Sample collection was undertaken in each well in lengths of 1 m or at smaller intervals if a sediment change was noted. The findings are outlined in Figures 4.1 and 4.2. Following casing installation, a down-hole perforator constructed for the purpose by McMillans Water Wells Ltd. (Figure 4.3) and driven by compressed air, was used to slot holes in the in-situ casing. This was the first time a slotter of this type has been used in Canterbury. Slot positions and the resulting casing porosity are given in Figures 4.1 and 4.2. Johnson Div. (1975) and Driscoll (1986) discourage the practice of casing perforation listing the following limitations on the method:

1. openings cannot be closely spaced;
2. openings are inaccurate and vary in size;
3. the percentage of open area is low; and
4. openings small enough to control fine to medium sand are difficult or impossible to produce.

Whilst these are pertinent factors, perforations resulting from the McMillan's slotter were much more controlled than suggested by Johnson Div. (1975) (Figure 4.4), and the method is endorsed in Woessner (1988) for the construction of monitoring wells. Compressed air was used to remove the punched divots as well as to develop the completed wells by jetting. Slotting was initiated at a depth of 7.75 m, below both the refuse base and the underlying silty sediment at the P.C.C. Landfill site.

Two 104 mm (4.6 inch) internal diameter observation wells were installed, approximately 10 m and 20 m downgradient from the pumped well. These were initially formed by installing a 6 inch casing using the driven air rotary technique, with the casing pulled back and removed from a 104 mm (4.6 inch) slotted PVC screen (Figure 4.5). In all four wells (M36/4018, M36/4120, M35/6103, M35/6218), a gravel

pack was not used; rather the surrounding sediment was allowed to collapse back into the annulus. The position of both the screen sections and the joints are shown in Figure 4.5. A fourth well, M36/4302, was emplaced at the northern end of the P.C.C. Landfill site to assess the degree of site variability and to collect sediment and refuse samples (Figure 4.6). Installed by driven air rotary, the steel cased 150 mm (6.6 inch) well was neither slotted nor screened, but was left open at its completed depth of 16 m to act as a potentiometer and to allow water sampling. The positions of these wells, installed by McMillan Water Wells and finished with locked well caps, are shown on Figures 4.7 (Wigram) and 2.7(P.C.C. Landfill).

Drilling was into waste of an unknown form at the P.C.C. Landfill site and the development of methane was likely; hence several safety precautions were taken. Because steel well sections were both welded together and gas axed apart, a "sniffer" unit (Figure 4.8) was used in and around the site to detect explosive gases. Personnel wore protective gloves and overalls, and used disinfectant on hands and faces prior to meal breaks which were taken away from the drilling site. The possibility of non-explosive but toxic gases was minimised, both because the site was exposed to strong winds but also because the drill rig and personnel were always upwind of the borehole being driven.

4.2.2 MONITORING WELLS

The position and number of hydrochemical sampling / potentiometric monitoring wells was decided following a geophysical assessment of the P.C.C. Landfill site (outlined in Chapter 5). A simple design with 34 mm (1.5 inch) internal diameter steel water pipe was used. Typically a 2 inch internal diameter casing is considered the smallest suitable size for groundwater monitoring wells. The screen is emplaced into a larger diameter driven casing which is backfilled with a sand pack and removed. Hydrochemical sampling is then undertaken, often with the use of a small submersible pump. At the onset of this investigation, the smallest commercially available submersible pump was 3 inches in diameter, whilst smaller pumps were available from other sources but the cost was prohibitive to this investigation.

Driven wells were therefore constructed of 34 mm internal diameter steel water pipe with the basal 200 mm length drilled with 10 mm diameter holes to allow water inflow. All monitoring wells were installed by the Canterbury Regional Council and were

completed with screw caps just above ground level ensuring some security and unobtrusiveness. Monitoring wells were arranged in clusters to allow hydrochemical sampling at 1 m depth intervals from 4.5 to 8.5 m at the same location, although at several sites only one such well was installed. Where wells ended in silty sand which was not clearable, well depth was extended by one metre, followed by surging. Following installation, each well was surge developed and /or overpumped to clear infiltering fines; repeated and sustained pumping was carried out to purge wells for geochemical sampling via a surface pump and down-hole hose and suction foot. Lithologic sampling was not possible from these wells but the driven installation method ensured minimal sediment disturbance about the well casing. Positions of the final 16 monitoring wells are shown in Figures 2.7 and 4.9.

4.3 SAMPLE LOGGING AND ANALYSIS

4.3.1 LITHOLOGIES

Lithologic logs and size analyses are presented in Figures 4.1, 4.2, 4.5 and 4.6. Individual interval size analyses and methods are outlined in Appendix 4.1, whilst Figure 4.10 exhibits the size envelopes found for each of the sampled wells. At both sites, sediments were dominated by fine sands or fine to medium gravels.

At Wigram, the full length of the installed wells penetrated gravels and sandy gravels (Figures 4.1, 4.2 and 4.5). Clays when encountered were found as small pods within and cementing gravels. If these formed distinct thin aquitards their significance was lost in the gross sampling method employed. Wood obtained at a depth of 13.67 to 14.17 m was carbon dated at 6660 ± 60 years B.P., approximately the time of the maximum seaward incursion throughout Canterbury during the Holocene. Clay and wood were also noted in M35/6103 between 15 and 17 m, but were absent in M35/6218. The localised extent of this deposit is probably a function of cross cutting alluvium channels. An exposed example of this was noted in the quarry wall of the hardfill tip on Owaka Road (Figure 4.11) where the 0.5 m thick unit is only 2 m wide. Where subsurface techniques are used to define the lateral extent of fine aquitard units pod shaped rather than sheet-shaped units are likely to be encountered.

At the P.C.C. Landfill, the content of refuse was varied and included paper, plastic, glass cloth, bone, metal, wood and pottery. A large percentage of the collected material included sand and gravel but mixed with unidentifiable putrescible refuse, ash

and black sludge. In M36/4017, pieces of metal and plastic wire were removed from the sand bailer when it was extracting sediment from a depth of 8.63 to 8.93 m, whilst a piece of can was removed during bailing at the 6.93 to 7.63 m interval. This refuse is thought to have been entrapped in the bailer from previous sampling levels. The refuse base was established at a depth of 5.93 m, immediately underlain by a blue silt (muddy sandy gravel) unit approximately 1.5 m thick. Gravels and sandy gravels throughout the remaining well depth are only interrupted by a silt and clay rich unit between 12.79 and 14.4 m. Twigs of Kanuka and Totara (Figure 4.12) returned a radiocarbon age of 8820 ± 70 years B.P. Similar lithologies were obtained from wells M36/4018 and M36/4120.

In well M36/4302 at the northern end of the P.C.C. Landfill, refuse was immediately underlain by sandy gravels and gravels above a silty sand encountered at 8 m. The silt and clay fraction of sediment increased progressively with depth to a maximum between 11 and 14.3 m, beneath which gravels once again dominated. In contrast to the localised distribution of the fine sediment at Wigram, this silty sand unit was encountered in monitoring wells throughout the site and is thought to completely underlie the landfill (Figure 4.9), although as drilling indicates, its thickness is variable.

Falling head permeability tests were undertaken on silty sand samples collected from both M36/4017 and M36/4302, with the results tabulated in Table 4.1. Values range from 1.49×10^{-3} m/day to 2.53×10^{-4} m/day in M36/4302, whilst a value of 2.80×10^{-2} m/day was obtained from the 5.93 to 6.93 m depth in M36/4017.

4.3.2 BOREHOLE GEOPHYSICS

Wireline geophysical logging was undertaken by DSIR Geophysics Division in two wells at Wigram, M35/6102 (Figure 4.13), M35/6103 (Figure 4.14), and one well at the P.C.C. Landfill, M36/4017 (Figure 4.15), with the following intentions:

1. to detect any aquitard layers overlooked by the method of lithologic logging and sampling;
2. to characterise sediment porosity and permeability; and
3. to detect the presence of any leachate developing in the location of bores at the P.C.C. Landfill site. (Chloride and boron both found at high levels in leachate, have high neutron capture cross-sections (Scott Keys & MacCary 1971, Ellis 1987) and should show up on geophysical plots if

present).

Because wells were cased, only the nuclear suite of neutron, natural gamma and gamma-gamma (Density-1 and Density-2) logs were run along with caliper logs. Calibration curves for the types of well casing used in this investigation were unavailable and the recorded counts per second (c.p.s.) could not be converted into environmental properties such as porosity. Trends and not quantified values are therefore presented.

The caliper logs of wells M36/4017 and M35/6102 shows an internal well diameter ranging between 6 and 6.4 inches, increasing as a result of the positioning of casing slots. In M36/4016 and M35/6102, the caliper value decreases rapidly at the base of the well, whilst values on the Density-2 (γ-γ) plot rapidly increase. This is in response to the respective sondes meeting a turbid zone at the base of wells produced by the settlement of infiltering fines and is not a function of the aquifer sediment or casing properties. In the PVC cased well, M35/6103, a sudden decrease in the well diameter is recorded circa 15 m. This is not coincident with any joints and as the only bore constriction observed is anomalous.

Both wells at Wigram display relatively featureless plots and therefore contain quite homogenous sediment. The water table is clearly delineated on Density-1 (γ-γ) and neutron logs in each case. A small Density-1, Density-2 and neutron trough at a depth of 15 m is representative of increasing clay and decreasing porosity content. This is coincident with a clay-rich gravel unit in M35/6102 found at a depth between 14.17 and 14.47 m and recorded between 15 and 17 m in M35/6103. This thin aquitard layer is absent from the lithologic log of M36/6218, and appears to be limited in extent.

The Density-2 (γ-γ) plot has the recorder closer to the emittance source and is more affected by casing properties. On both the M35/6102 and M36/4017 Density-2 (γ-γ) plots, decreased c.p.s. coincide with welded casing joints between slotted casing. The effect is not noted on other nuclear logs nor in the M35/6103 well where PVC casing was installed.

Between 13 and 14.5 m in well M36/4017 at the P.C.C. Landfill, an increase in Density-1 and Density-2 c.p.s. and a decrease in neutron values indicates a unit of

higher clay content and lower porosity detected in lithologic logging between 12.79 and 15 m. The water table which was positioned at the time of logging at a depth of 5.11 m is clearly delineated, but the silty sand unit immediately underlying the refuse is not clearly defined on the geophysical log. The natural gamma c.p.s. increase and the decrease in the neutron count above the water table in M36/4017 may result from a perched water table within refuse at a depth of 3.2 m. No leachate contaminant zone was evident from down-hole logging in this well at the P.C.C. Landfill.

4.3.3 WATER LEVELS

Seasonal water levels within aquifer sediment deeper than 7.5 m were monitored by automatic water level recorders in wells M36/4018 at the P.C.C. Landfill, and M35/6103 at Wigram. Both sites exhibit very regular seasonal fluctuations on the order of 0.9 to 1 m at Wigram and 0.7 m at the P.C.C. Landfill, from summer highs to winter lows. Weekly water level averages for the two study locations are shown in Figures 4.16 and 4.17. Aquifer recharge through the months of May to September give rise to increasing water levels, with values highest over the early summer months, and rapidly declining during late summer, January to February in 1989/90, November to February in 1990/91 and October onward in 1991/92. The seasonal pattern is the reverse of rainfall records and results from a lag effect in aquifer recharge. The two site records are very similar indicating that the same aquifer is being monitored at each site.

Unlike Wigram however, the P.C.C. Landfill overlies a shallow aquitard layer. Long term monitoring above this silty sand unit was not possible with potentiometer wells only recently installed; however water level relationships, as recorded on 4/10/91, are given in Table 4.2. Levels were recorded using a hand-held water level probe, with errors of ± 5 mm on each reading.

In the northwest well cluster, M36/4317 ended in gravels below refuse at a depth of 6.5 m, and had a piezometric water level of 15.509 m above mean sea level (a.m.s.l.). Wells M36/4319 at 7.5 m deep and M36/4320 at 8.5 m deep, which end in silty sand, have water levels of 15.410 and 15.335 m a.m.s.l., respectively. Water in M36/4302, set to a depth of 16 m, registered at 15.349 m a.m.s.l. Water in the shallowest well, M36/4317, is therefore perched above the silty sand aquitard and downward leakage potential exists. Similar results are evident from other well clusters. Between

piezometric monitoring wells M36/4317 and M36/4327, both sited at the base of refuse but above the silty sand, a hydraulic gradient of 5 x 10^{-4} m/m oriented at a true compass bearing of 155° exists. Well M36/4309 exhibited a static level of 16.178 m a.m.s.l., higher than other levels recorded. Whether this is a result of water table mounding at the site or due to a perched water table within the refuse pile is unknown.

The full three-dimensional hydrogeologic connection between the other water level measurement points is unknown and cannot be used to define a groundwater table contour map (cf. Saines 1981). However, clearly the area of refuse deposition at the P.C.C. Landfill site is basally confined and hydrologically distinct from the aquifer used to characterize the regional flow pattern in the reconnaissance survey. The relationship between the units is undefined and it has not been established whether the shallow water table is percolation or groundwater recharge derived. The contribution to downward percolation from irrigation sprinklers on the greyhound race track, which are used during the summer months to soften the track surface for racing, should be considered in a more thorough water balance of the site.

4.4 PUMP TESTS

Constant discharge rate pump tests were carried out at both the P.C.C. Landfill and the Wigram sites in order to evaluate gross aquifer hydraulic properties. Data collected during the two pump tests and curves used in the test analysis are given in Appendix 4.2. For analysis methods the reader is referred to Kruseman & DeRidder (1970). Results using a variety of analysis methods are given in Table 4.3.

4.4.1 PAPARUA COUNTY COUNCIL LANDFILL

The first test at the P.C.C. Landfill (28/3/90) utilised pumped well M36/4017 and observation bores M36/4018 and M36/4120 which were slotted at depths of 7.5 to 30 m below the landfill base. A four inch submersible pump, set at a depth of approximately 15 m below the surface in M36/4017, discharged water at the maximum sustainable rate of 19.51 l/s through a 100 mm orifice plate at the surface, and was used to evaluate the flow rate.

Two limitations affected the P.C.C. Landfill pump test. The aquifer was clearly capable of delivering far more water than the pump which meant a longer period at

the lower discharge rate was required to achieve a steady state drawdown cone. In addition the water had to be delivered away from the immediate site to prevent percolation back into the underlying aquifer. Water was discharged onto the southern topographically lower end of the P.C.C. Landfill site. During the test this discharged water ponded and began to overflow across the nearby road intersection. To prevent flooding, pumping was stopped after 24 hours. In Canterbury, the phenomenon of delayed yield in unconfined aquifers is a rarity, but had it occurred in the P.C.C. Landfill pump test it would have been noted by inflection of the drawdown curve circa 29 hours. The pumping interval tested however is confined. The degree of water ponding at the southern section of the site indicated fine impermeable surface sediment and well logs further north show the existence of a silty sand immediately below the landfill base. The confining nature of the unit was confirmed by the storage coefficient evaluated from the pump test (Table 4.3), which averaged 9.59×10^{-4} for observation well M36/4018 and 8.78×10^{-4} for observation well M36/4120.

Excluding the Theis Recovery Method of analysis for which transmissivity (T) results are a factor of 2 to 3 greater than values found from other methods, T averages 3.38 x 10^3 m^2/day (3.9×10^{-2} m^2/s) for M36/4018 and 5.9×10^3 m^2/day (6.8×10^{-2} m^2/s) for M36/4120 giving a site average of 4.6×10^3 m^2/day (5.4×10^{-2} m^2/s). These values are similar to that calculated from Logans Approximation of 3.46×10^3 m^2/day. Callander (1988) established similar transmissivity values for two bores drawing from different aquifers in the Wigram RNZAF Base water supply group: 2.7×10^{-2} m^2/s for a bore set at 71 m below ground level, and 6.9×10^{-3} m^2/s for a bore 114 m deep.

Transmissivities are a more realistic comparative term for hydrogeologic parameters in the Canterbury setting when aquifer and aquitard units are ill-defined and the depth to true basement is several hundred metres (Thorpe 1990). However, if the length of the pumped well screen (22.5 m) is taken as the zone exclusively affected by the pumping test i.e., the aquifer depth, then the averaged hydraulic conductivity is calculated using the averaged site transmissivity value (2.07×10^2 m/day or 2.39×10^{-3} m/s). Thorpe *et al.* (1977) found near surface gravel porosities from field density tests on Heretaunga Plains alluvium, to range from 0.2 to 0.35. These gravels were similar to sediment at the Wigram and P.C.C. Landfill localities and a 30% porosity is assumed for these sites. The horizontal seepage velocity can then be derived using the local hydraulic gradient (8.9×10^{-4} m/m) (Chapter 3), as 7.09×10^{-6} m/s (0.61 m/day)

after Darcy's Law. The results are consistent with clean sand / gravel values quoted in Freeze & Cherry (1979).

4.4.2 WIGRAM

The second pump test at Wigram (2/10/90) utilised pumped well M35/6102 and observation wells M35/6103 and M35/6218 (Figure 4.7). A four inch submersible pump set at a depth of approximately 12 m discharged water through a 75 mm orifice plate and was used to evaluate the flow rate. Water was discharged at a rate of 15.7 l/s and continued for 48 hours. Delayed yield was not observed. Water level observations were affected to a minor degree by recharge interference. At the conclusion of the test, the water level in each of the observation wells was higher than the level prior to the test. Measuring errors of ± 5 mm account for some water level variation. However, a line drawn between the commencing and concluding water levels parallels the gradual monitoring change observed throughout the test (Appendix 4.2). This can be attributed to a natural water level increase over the duration of the test. Following the initial rapid level change up to 200 minutes, this recharge was significant enough to affect the drawdown curve in the pumped well, and in observation wells because the absolute changes were very small.

Transmissivity calculations were affected by recharge, but values for the two observation wells M35/6103 and M35/6218 were established using the Chow method (4.58×10^4 m^2/day and 3.97×10^4 m^2/day respectively), and by the Theis Recovery Method (1.77×10^4 m^2/day and 2.07×10^4 m^2/day respectively, Table 4.3). Flattened drawdown curves made curve fitting methods difficult: however, a Boulton transmissivity value of 2.46×10^4 m^2/day and Walton transmissivity value of 1.50×10^4 m^2/day were obtained for the second of the wells (M35/6218). Logans Approximation of 2.72×10^4 m^2/day is a magnitude smaller than calculated from other analysis methods for the Wigram pump test. Excluding this value, the results from the other analysis methods are averaged to give a transmissivity value representative of the site, 2.73×10^4 m^2/day. Hydraulic conductivity is therefore calculated as 1.21×10^3 m/day (1.4×10^{-2}m/s). Again assuming a volumetric porosity of 0.3 and the hydraulic gradient of 8.9×10^{-4} m/m, the averaged seepage velocity is 3.59 m/day (4.2×10^5 m/s).

A 30 cm thick silt-rich unit at 14.2 m was the only fine unit located during lithologic

logging. Either this unit alone or the varying hydraulic properties of interlayered sediments was sufficient that the pumped interval registers a mean storage coefficient of 4.52×10^{-3} which is representative of semi-confined conditions.

4.5 CONTAMINANT TRANSPORT CONTROLS

4.5.1 DISPERSIVITY

Contaminant flow has been conceived as being governed by the advection-dispersion equation. For one dimension in saturated, homogeneous, isotropic materials in steady state flow, this is given by:

$$\frac{dC}{dt} = -v'\frac{dC}{dx} + D\frac{d^2C}{dx^2}$$

where C is the solute concentration, v' is the average linear (seepage) velocity, x is the distance along the flow path from the contaminant origin, t is the time since the start of the contaminant invasion, and D is the dispersion coefficient. This is the simplest, conservative, case and does not include the effects of chemical variations, biological transforms or radioactive decay. The first term on the right relates to the concentration change with time brought about by the average groundwater flow, whilst the second relates to the microscopic processes affecting the solute mass. These effects are grouped under the term hydrodynamic dispersion (D) and were initially observed in the electrolyte experiments by Slichter (1902).

Slichter found that the concentration of a tracer detected in a downgradient observation well gradually increased to a maximum peak. In addition the tracer advanced in a pear shape, becoming wider and longer in the direction of flow. This spreading was thought to be governed by two processes contributing to the hydrodynamic dispersion:

$$D = \alpha_L v' + D^*$$

where D^* is molecular diffusion and α_L is mechanical dispersivity. Molecular diffusion is caused by random movement of molecules in a fluid and acts even when the fluid is static. An analogy is the spreading out, and diluting, of an ink drop released into a container of water. Diffusion is significant at low velocities, where $D = D^*$ (Knight 1986), but more often diffusion becomes negligible in comparison to mechanical dispersivity. Mechanical dispersivity results from velocity variations within pores due to sidewall effects, flow path tortuosity and pore diameter. It is maximal in the

direction of groundwater flow (α_L) and several orders of magnitude smaller in orthogonal (transverse) directions (α_T).

Both longitudinal and transverse dispersivities have been taken as an intrinsic property of the porous media representing a characteristic mixing length. Consequently the solution of the advection-dispersion equation and the characterisation of α_L and α_T for various aquifers and sediment types has dominated groundwater contaminant studies since the early seventies (Molz *et al.* 1989).

However, it became apparent from tracer experiments that dispersivity values were scale-dependent, with values increasing with the scale of measurement (Pickens & Grisak 1981, Gelhar 1985, Molz *et al.* 1983). Laboratory-determined values therefore bore little relationship to field based results, and several orders of field experiment were defined (Fried 1975). Further to this, both field and theoretical research found that longitudinal dispersivity did not continually increase with the experimental scale but reached an asymptotic limit, whilst transverse dispersivity ceased to have an effect on tracer spreading a short distance from the contaminant source (Lehr 1988). The concept of dispersivity was therefore poorly defined, and longitudinal hydrodynamic dispersion was found to be relatively unimportant in the transport of contaminants (Molz *et al.* 1989, Lehr 1988).

The dispersivity scale effect arises because of the presence of aquifer heterogeneities. Larger dispersivity values were therefore found in experiments using long well intake screens because the contaminant was influenced by a large number of heterogeneities, whilst smaller dispersivities were established using point samples because the sediment was more homogeneous over the small test interval (Pickens & Grisak 1981, Killey & Moltyaner 1988). Thus, the characterization of vertically-distributed heterogeneities in the form of hydraulic conductivity has been recognised as fundamental to the understanding of contaminant flow behaviour. Molz *et al.* (1989) view this move from vertically-averaged to vertically-distributed parameters as "the next step" in the understanding of contaminant transport. Numerous "next step" techniques have recently been developed which are examined in the following section.

4.5.2 HYDRAULIC CONDUCTIVITY

Taylor *et al.* (1990b) evaluated six methods for determining the vertical distribution

of hydraulic conductivity including the established approaches of: particle size methods, empirical relationships between electrical and hydraulic conductivity, and straddle packer tests. Three newly developing techniques presented and discussed were: Stonely Wave attenuation in the use of the sonic logging tool, the single well electrical tracer (SWET) test (wherein the passage of a salt tracer introduced the full length of a slotted well casing is monitored by an induction tool), and hydraulic conductivity calculation based on the flow of groundwater through fill material emplaced in the well. Other techniques being developed and refined include the use of impeller flowmeters (Molz *et al.* 1989b), a heat-pulse flowmeter (Kerfoot & Kiely 1989, Kerfoot 1988, Guthrie 1986), and a resistivity borehole dilution device which uses a azimuthal Wenner electrode arrangement about a solid core to detect the direction and velocity of a salt tracer's passage from a well (McLinn & Palmer 1989). In addition, an electronic cone penetrometer, which assesses the resistivity of material at its driving tip, is finding use in some settings (Campella and Weemees 1990). Many of these techniques are simply beyond the operational constraints of this investigation. However, the two simplest traditional techniques are examined in more detail in the following sections.

4.5.2.1 Particle Size Methods

In unconsolidated deposits, hydraulic conductivity is a function of both fluid properties and the intrinsic properties of the aquifer media, including grain size distribution, particle roundness and packing. The relationship is commonly reported as:

$$K = \frac{\rho . g}{\mu} . C d^2$$

where K is hydraulic conductivity, μ is fluid viscosity, ρ is fluid density, g is gravitational acceleration, C is a constant representing the physical characteristics of the flow media, and d is the media particle size parameter.

Hazen (1893) initiated the use of the form $K = A d_{10}^2$ where the size parameter was established on a weight basis, where 10% of the sediment was finer than the size stipulated. Masch & Denny (1966) took 'd' to be the inclusive standard deviation of samples as plotted on a size gradation curve. Other researchers have refined or amended these predictive equations for K, usually on the basis of results from laboratory flow studies using uniform spherical grains. The results are therefore

limited to predicting the behaviour of well sorted sand deposits (Taylor *et al.* 1987, Taylor 1990b). These sediments can be easily characterised by single parameters derived from size gradation plots as they have well defined 'S' shaped distributions. Silts and clays are unsuitable for sieving, and small sedimentary features which are lost in the sampling process have a more significant effect on the hydraulic conductivity of these sediments. However, silt deposits and mixed size sediments, including river alluvium, were incorporated in an assessment of 19 published data sets, of both real and synthetic materials, by Shepherd (1989) to show that whilst the $K = C\ d^x$ relationship was applicable for unconsolidated clastic sediments, both the constant and exponent decreased with textural maturity and increased with induration. A example of this is provided by Uma *et al.* (1989) and Egboka & Uma (1986) who found that sediments from each of three sand aquifers in Nigeria were internally consistent with regard to the size versus hydraulic conductivity relationship but that the aquifers could be distinguished by different constant values which was attributable to the nature of the geologic environment, i.e., to texture and induration.

Gravelly deposits are problematic in this approach both because it is difficult to collect the large clasts and ensure a representative sample is obtained, and because gravels tend to have elongate drawn-out tails on a size gradation curve and are therefore less easily represented by a single parameter unless it includes some rating of each section of the curve. On this basis, size correlation with hydraulic conductivity is usually considered unworkable. However, a substantial volume of sediment was collected for each logged interval in wells M36/4017, M36/4302 and M35/6102, thereby satisfying the representative sample requirement, and the only restriction on clast collection was the bore diameter. Only twice during drilling was a 'goolie' (a clast bigger than the bore diameter) encountered which had to be broken before drilling recommenced. Size analysis results given in Appendix 4.1 are shown in Figures 4.1, 4.2 and 4.6, along with their calculated uniformity coefficients (d_{60}/d_{10}). Whilst a laboratory based correlation between size and hydraulic conductivity is not attempted, size analyses will be compared with field assessments of hydraulic conductivity determined by the second technique, point dilution packer tests, outlined in the following section.

4.5.3 STRADDLE PACKER / POINT DILUTION TESTS

4.5.3.1 Method and Materials

The point dilution technique was devised in the USSR in the 1940's (Freeze & Cherry

1979) and has become a standard means of determining the hydraulic conductivity of sectioned-off portions of a screened borehole, as outlined in Appendix 4.3. The method uses a single tracer pulse in a well section to identify high velocity, and therefore preferential contaminant transport zones, within an aquifer. Its use is limited in settings where high vertical flows exist, or significant leakage occurs in the disturbed sediment or gravel pack annulus around the well casing. The first of these limitations is not applicable at the P.C.C. Landfill with all levels to be tested in the same aquifer, whilst the second has been minimised by the well installation techniques (section 4.1).

Initially radioisotopes were used as the diluting solute (Drost & Neumaier 1974, Halevy *et al.* 1967, Gaspar & Oncescu 1972), but the advent of specific ion electrodes and portable conductivity probes and meters has meant the technique is now cheaper and more convenient to attempt (Grisak *et al.* 1977, Freeze & Cherry 1979).

One of the two models of point dilution packer apparatus (or straddle packer) designed by the author and Technician Arthur Nicholas, and used in this investigation is illustrated in Figure 4.18. Although conceived and built without reference to other multipacker systems, the equipment (as with many simple concepts) is by no means unique. A very similar system is featured in Barczewski & Marschall (1989) where the separate packer intervals are used for depth integrated water sampling using either suction or miniature submersible pumps. When used for this purpose Taylor *et al.* (1990a) warn that sampling may be unduly affected by water drawn into the sampling section from above and below the packers unless seals are emplaced between the casing and the formation at the ends of sample intervals. In that same study, however, the straddle packer was successfully used for point dilution tests using the dilution of fresh water by contaminated water in the aquifer to delineate pore fluid stratification. Straddle packers can also be adapted for use as vertically discrete observation points in two well tracer tests (Melville *et al.* 1988).

4.5.3.2 Reproducibility and Error Sources

Point dilution tests were initiated with trials to refine equipment handling, operation and observation recording. Two trials in each of three well sections were undertaken in well M36/4017. In each case, tracer injection continued until several consecutive conductivity values reached a plateau value where tracer inputs and outputs were

steady. Thereafter tracer pumping was discontinued and the conductivity decline with time was recorded. The results of these reproducibility tests are illustrated in Figure 4.19.

The decline in salt concentration with time for the two trials in each of the three intervals are remarkably similar despite the fact that the tracer injection rate varied, with plateau values different in each case. Thus, a standardised pumping rate and volume is not required as the dilution slope is reproducible.

Pumping rate and volume significantly affect other test factors which were examined in a second method trial (Figure 4.20). A high pumping rate supersedes the horizontal flow velocity, forcing water laterally into the surrounding sediment. Not only does it take longer for the centre of mass of the solute slug to migrate from the detection zone as a larger volume has been emplaced, i.e., a longer time is spent at the plateau value prior to dilution, but the tracer forced upgradient of the sectioned-off interval travels back through both this section and the underlying interval. This causes conductivities in the lower (L) interval to increase as the tracer is introduced into the middle (M) interval (exemplified in Trial 3 in Figure 4.20), negating the function of the conductivity probe in the lower (L) section which was to detect downward tracer migration via leakage along the outside of the casing. This effect is minimised if the injection rate is slower (e.g., Trial 1 in Figure 4.20) or a vertically longer packer is used. It is possible when the tracer is forced into both intervals by this 'flowback' mechanism, that the conductivity decline in each interval can be used to calculate their respective seepage velocities. In Figure 4.20, Trials 1 and 3 show the same replication of dilution curves in the lower interval as in the middle injection interval. In Trial 2 (Figure 4.20), gravity rather than pumped tracer injection introduced an insufficient tracer volume to the M interval to ensure solution homogenisation but the slight conductivity rise in the lower section is sufficient to show that vertical leakage along the casing annulus is minimal in comparison the lateral movement of the tracer.

In Figure 4.19, conductivity values recorded in the injection phase of trial 2 for the 14.44 to 14.94 m interval are highly erratic in response to an unstable electrode connection. In Figure 4.19, tracer dilution was rapid in both trials in the 20.94 to 21.44 m interval, and few data points were recorded on the descending curve for velocity calculations. Instrument errors are ± 0.1 mS/cm, but because values were hand

recorded at 10 second intervals, errors of up to 1 mS/cm and ± 1 s affect the accuracy of subsequent calculations for the rapidly diluting intervals. With increased method familiarity, some of these handling and recording problems were ameliorated.

4.5.3.3 Results

Results from point dilutions in M36/4120 are given in Figure 4.21, whilst those performed in M36/4017 are illustrated in Figure 4.22. In all tests undertaken there was no variation in the conductivities measured within the section packed off above the injection interval. Thus, only results for the middle (M) injection interval and the lower (L) section are shown.

Several handling and equipment problems were experienced which are reflected in these figures. With use, the plug connections from the coaxial cable to the conductivity meters loosened and there was difficulty getting consistent readings in some tests. This is shown in the excessively noisy conductivity plots for the injection interval (M) in M36/4120 in Figures 4.21q, 4.21r, 4.21s, and the sudden jumps in conductivity values (Figure 4.21f), and in the lower interval of trials undertaken in M36/4107 (Figure 4.22b, 4.22l, 4.22m, 4.22i, 4.22g). In tests shown in Figure 4.21l and 4.22o, pumping was discontinued and there were handling problems experienced with the equipment. Salt injection was therefore restarted, and pumping was discontinued following a second plateau. In Figure 4.21l, the second plateau is at a higher concentration than the first, whilst in Figure 4.22o the converse applies. In the test illustrated in Figure 4.22l, salt water seeped into the casing of the conductivity meter, shorting out the circuit board and making the meter inoperable. Hence for that test only the results from the injection interval are shown.

Point dilution curves were analyzed (as outlined in Appendix 4.3) to calculate the horizontal average linear velocity within each of the injection intervals. For the 7.44 to 7.94 m interval in M36/4017 (Figure 4.22a), tracer dilution over the 2.5 hours of the test was almost imperceptible. The slight changes were within the error margin of each reading and hence could not be considered significant enough for use in velocity calculations. The results for the other point dilution tests are listed in Table 4.4 and shown in Figure 4.23.

Within well M36/4120, horizontal average linear velocities ranged over three orders

of magnitude, from 0.072 m/day (8.3×10^{-7}m/s) to 59.75 m/day (6.9×10^{-4}m/s) with an overall bore average of 11.39 m/day (1.32×10^{-4}m/s). Within M36/4017, horizontal average linear velocities ranged over four orders of magnitude, from 0.088 m/day (1.01×10^{-6}m/s) to 237.6 m/day (2.75×10^{-3}m/s) and averaged 61.9 m/day (7.14×10^{-4}m/s). Whilst similar low velocity values were found for both wells, higher values were found for well M36/4017. Further work is required to ascertain if well construction factors give rise to this feature, or if this merely reflects sediment variability across the site.

Assuming that a porosity of 0.3, and the regional hydraulic gradient of 8.9×10^{-4} m/m applies to each of the test intervals, the hydraulic conductivities are calculated (after Darcy's Law) for the wells as follows: M36/4120 ranges from 4.4×10^{-2} m/s to 2.4×10^{-1} m/s; and M36/4017 ranges from 2.3×10^{-1} m/s to 9.2×10^{-1} m/s. These values are consistent with the silty sand to gravel range outlined in Freeze & Cherry (1979), and a factor of ten higher than the hydraulic conductivity value (2.39×10^{-3} m/s) established from the site in the pump test (Section 4.4.1). Given the errors and assumption involved in methods and calculations, the results are seen to be reasonably consistent.

Though the two test wells penetrate braided fluvial deposits, they are sited only 10 m apart, and the point dilution results show a strong correlation between the bores. Three zones of higher velocity are defined, each separated by low velocity layers. The low velocity of 8.8×10^{-2} m/day (1.01×10^{-6} m/s) determined in the 13.44 to 13.94 m interval in M36/4017, is coincident with a muddy sand unit containing 5% clay and 10% silt. This unit was also encountered in well M36/4120 where it was recorded as a blue silt (Figure 4.5). Falling head permeameter tests on this sediment from the 5.93 to 6.93 m interval in well M36/4017 resulted in hydraulic conductivity values of 2.8×10^{-3} m/day (3.24×10^{-7}m/s). Hydraulic conductivity values ranging from 2.53×10^{-4} m/day to 1.49×10^{-3} m/day (2.9×10^{-9} m/s to 1.73×10^{-8} m/s) were established from falling head permeameter tests on blue silty sand samples from well M36/4302. Similar silt-rich sediment was responsible for the extremely slow tracer dilution in the 7.44 to 7.94 m interval in M36/4017 (Figure 4.22a).

Fluid percolating downward would therefore travel very slowly through the silty sand unit and would be laterally carried from the site through the more permeable gravel and sandy gravel sediments, presumably carrying with it any contaminants which had

not been filtered out or adsorbed in the silty sand.

The low velocity readings for the intervals between 16.44 and 19 m in well M36/4017 is coincident with a zone where sampling problems were encountered during drilling. Sediment heaving, where the sediment resupplied the sample interval faster than the sand bailer could remove it due to pressure release, occurred at this depth. It is possible that heaving resulted in the under-representation of fines in this interval in the logged and collected sample, although some clay pods were extracted with gravels.

Despite the acknowledged problems of correlating the vertically-distributed hydraulic behaviour of coarse or fine grained sediments to their size properties, seepage velocities established from point dilution tests in wells M36/4017 and M36/4120 show similar patterns to the uniformity coefficients of the lithologies (Figure 4.24). No attempt to describe the relationship in terms of $K = Cd^x$ was undertaken because point dilution test intervals did not coincide with sampling intervals, but high seepage velocity zones overlap areas of sediment with high uniformity coefficients.

4.6 SUMMARY

Wells were installed at both the Wigram and P.C.C. Landfill investigation sites to more clearly define the local hydrogeology and to gain access to the subsurface for tracer tests characterising vertically-distributed properties. At Wigram, lithologic logging, sampling and borehole geophysics reveal the sediment to be relatively homogeneous, consisting of fine to medium gravels and sandy gravels with a gross transmissivity value of 2.72×10^4 m^2/day, a hydraulic conductivity of 1.21×10^3 m/day, and a storage coefficient of 4.52×10^{-3}.

Laterally restricted clayey gravels were found at a depth of approximately 14 m with wood samples dated at 6660 ± 60 years, the date of the most landward sea incursion known throughout Canterbury during the Holocene. Similar Kanuka and Totara samples from a comparable depth at the P.C.C. Landfill were dated at 8820 ± 70 years.

Drilling, and lithologic and geophysical logging at the Paparua County Council Landfill revealed a silty sand unit, which although varying in thickness, is found throughout the site and acts as a basal confining aquitard, with water perched above it saturating the

base of overlying refuse. The hydraulic conductivity of this sediment was established from falling head permeameter tests at between 2.53 x 10^{-4} and 2.80 x 10^{-2} m/day, considerable lower than the 2.06 x 10^{2} m/day calculated for the underlying sandy gravels and sands from a standard pump test. A storage coefficient of 9 x 10^{-4} and transmissivity of 4.62 x 10^{3} m^2/day were calculated for this locality.

A borehole dilution straddle packer system was designed and built for the investigation of vertically-distributed permeability characteristics below both refuse and silty sand units at the P.C.C. Landfill. This low cost and easily used equipment was found to give consistent, reproducible results, despite minor operating problems. Hydraulic conductivities were established by this technique at between 4.4 x 10^{-2} and 2.4 x 10^{-1} m/s in well M36/4120, and between 2.3 x 10^{-1} and 9.2 x 10^{-1} m/s in well M36/4017. Not only were distinctive high permeability zones differentiated within the wells, but these zones could be correlated between the two wells, thus indicating laterally continuous lithologies which could provide possible preferred transport paths for leachate contamination beneath the landfill. Seepage velocities also showed a strong correlation with the uniformity coefficients calculated from sediment size analysis results, despite the recognised difficulties of size representation in very fine and coarse sediments.

The downward percolation of fluids from the landfill will therefore be inhibited by the low permeability of the silty sand unit. Contaminants which are retained in leachate below this unit may be carried laterally from the site in the higher seepage velocity zones detected in the underlying gravels and sands. However, this depends both on the presence and physical properties of the contaminants themselves which are, at this stage of the site investigation, undefined.

CHAPTER 5 SURFACE GEOPHYSICS

Throughout this text it is assumed that the reader is familiar with a basic knowledge of geophysical terms and methods including: direct electrical resistivity, self-potential, Wenner, VLF, Schlumberger, dipole-dipole arrays, formation factor, conductivity, seismic refraction and borehole logging techniques. General details of these can be found in Parasnis (1979), Sharma (1976), Dobrin (1976), Griffiths & King (1979) and Telford *et al.* (1976). Specific applications of geophysical methods to groundwater studies are found in Zohdy *et al.* (1974) and Fetter (1988), with electrical resistivity use in Bhattacharya & Patra (1968) and Koefoed (1979). These standard definitions and method outlines are omitted here because they are well detailed in the named works. Concepts which require additional explanation in line with their use in this project are outlined in the Appendices.

5.1 INTRODUCTION

Groundwater contaminant investigators fulfil the role of Little Jack Horner; only on rare occasions are they able to directly access their resource, and rarer still strike a plum location. Rather than plunge in with time consuming and expensive "hit and miss" drilling (or trenching), noninvasive surface geophysical techniques can be used in the reconnaissance of existent, natural or anthropogenic contamination, prior to planning and undertaking groundwater sampling (Greenhouse & Slaine 1986, Urish 1983, Stollar & Roux 1975). This approach is well established and documented (Zohdy *et al.* 1974, Fetter 1988), and has been applied in the United States and elsewhere for twenty four years (Cartwright & McComas 1968); it is rarely used in New Zealand (see Section 5.3.1). Geophysical methods can also provide an enhanced overall understanding of complex site conditions because of their spatial coverage.

A second technique uses the invasive release of an easily detectable artificial contaminant to reflect groundwater behaviour. Substance migration behaviour can then be traced at point sampling sites, or in some cases (as in this investigation), by surface geophysics (Fried 1975).

The combination of these methods should satisfy those geophysical technique capabilities as outlined by Mazac *et al.* (1987) to involve:

1. determination of site characteristics, hydrogeological, lithological or

structural;

2. assessment of land surface and groundwater relationships;
3. detection of pollution presence; and
4. estimation of pollutant migration patterns and concentration.

Some studies focus on contaminant behaviour in the aerated vadose zone (Frolich & Parke 1989), but most apply to the saturated zone, as in this investigation.

5.1.1 SITE SELECTION

When both existent and induced contaminant characterisation is desired, the best, and usual, approach is to undertake induced experiments on land constrained by the same hydrogeologic setting, but beyond the present and potential contaminant pathways, as exemplified by the Borden Landfill in Canada (Sudicky *et al.*, Dance & Reardon 1983). Thus, an induced contaminant is not superimposed onto the existing leachate plume, which would make the distinction between them difficult because each contaminant has an unknown effect on the other. In addition the introduced tracer does not become subject to complex physiochemical and biological reactions ongoing within the refuse and leachate. However, the Paparua County Council Landfill did not present this possibility, as land immediately to the west is an intensively used horticultural block, and land to the north (upgradient) is still undergoing gravel extraction and hardfilling. The areas to the south and east are only periodically sown in fodder crops and otherwise grazed, but are unsuitable, as the land is privately owned, and the potentiometric survey (Section 3.4.3) indicates leachate migration likely in this direction at an unknown depth.

A hydrogeologically similar site is the "Forty Acre Block" on the Wigram RNZAF Base which lies to the northeast of Carrs Road, occupying a parallel paleochannel of the Waimakariri River. This site is also accessible long term for borehole installation and monitoring, and surface geophysics. Forming part of a flying corridor not in grounduse, the permanently secured, flat well-groomed grassed area is unhindered by vegetation and cultural obstacles, and lies at least 1 km upgradient from pumped water supply wells.

The two field sites are as similar as possible given hydrogeological and geographic constraints, and whilst experiments at the Wigram site will not replicate contaminant behaviour at the P.C.C. Landfill, the findings should be generally applicable to that

location. The following surface geophysical investigations are thus presented in two sections: induced and existent contaminant studies, which equate to the two separate sites, Wigram for the former, the P.C.C. Landfill for the latter.

5.1.2 METHOD SELECTION

5.1.2.1 Tracer

Tracers have been used since biblical times to detect water migration pathways and to determine flow direction, velocity and dispersion characteristics. Substances used have included: stable and radioactive isotopes, yeast, spores, bacteria, dyes, sugars, gases, acids, paper and plastic floats, electrolytes and olfactory tracers such as skunk odour (Davis *et al.* 1980, 1985); some of these are obviously unsuitable for groundwater use. Successful tracer use is heavily dependent on the balance of inherent substance properties (Figure 5.1) and sociological, technical, and site factors (Figure 5.2). A salt electrolyte, NaCl, is chosen for use at Wigram in part because it enables simultaneous use of direct and indirect detection techniques.

Chloride ions typically exhibit rapidly moving, undistorted sharp elution curves when used as a tracer (Kaufman 1956 in Davis *et al.* 1985). Typically, as with other simple ionic tracers, Cl ions will not decompose and be lost to the system. In addition to being detectable using DC resistivity techniques, it can be simply monitored using conductivity probes at both the release point and at monitoring wells downgradient, particularly in settings where background conductivities are low, as in the Christchurch aquifers. This latter technique, first used by Slichter (1902), negates the need for the collection or laboratory analysis of water samples.

5.1.2.2 Surface Geophysics

The application of geophysical techniques to waste migration studies has been evaluated by Benson (1989) and Mazac *et al.* (1987) who found geoelectric methods - specifically galvanic resistivity (DC) and electromagnetic methods (EM), to be particularly useful and reliable for most contaminant applications, when used in appropriate hydrogeologic settings (G.S.E.G. 1988) . Both DC and EM readily respond to changes in fluid conductivity brought about by ionic loading of groundwater with the introduction of contaminant dissolved solids. Contaminated zones are detected as low resistivity (high fluid conductivity) anomalies in the geoelectrical patterns produced unless the anomalies are overridden by natural scatter, or there is

no contrast due to lithology changes. As with any geophysical technique, borehole data and water quality testing is used to assist and clarify the inherently nonunique results.

Selection of surface geophysical methods in this investigation is based on equipment availability, rather than time, cost or suitability basis. Of those methods appropriate for detection of induced or existing contaminants, only galvanic DC resistivity is readily available in New Zealand - a point also noted in Broadbent (1991). Suitable EM or VLF is available from Australia, but at a hirage and transport cost prohibitive to investigations of this scale. Similarly, continuous multicore vertical electrical sounding equipment, outlined in Van Overmeeren & Ritsema (1988), was unavailable. Traditional galvanic DC resistivity is therefore used exclusively in this investigation, except for a single seismic refraction profile undertaken at the Paparua County Council Landfill to assist site characterisation. Seismic refraction results are discussed in section 5.3.6.

It is hoped that other techniques (particularly EM) will become more readily available in this country with growing recognition of their speed and application. Instances of groundwater pollution occur in a variety of settings, not all suited to galvanic detection, and it would be advantageous to gain from both the combination of techniques and the use of alternate systems.

5.1.2.3 DC Resistivity Constraints

All groundwater contamination problems are unique, with individual cases controlled by specific site constraints that affect the geophysical methods chosen to tackle them (Mazac *et al.* 1989), as explored in case studies by Stollar & Roux (1975), Klefstad et al. (1975), Urish (1983), Hackbarth (1971), Greenhouse & Harris (1983), Greenhouse & Monier-Williams (1985) and Benson (1989). Site influences can be ameliorated in some instances, such as undertaking soundings perpendicular rather than parallel to sources of cultural noise. Often such noise can have disastrously uncompensatable effects on geophysical readings (as in Alao 1985). Only operator specific factors are controllable. In contaminant studies, detection methods are additionally influenced by the behaviour of the contaminant itself, never wholly predictable or controllable even in induced studies. Hence, the success of contaminant detection is never guaranteed. Thus, a stepwise procedure based on the concepts of Urish (1983) and

Stierman & Ruedisili (1988) is undertaken in this investigation (Figure 5.3).

At each stage, the site specific factors limiting the geophysical method are assessed and the feasibility of each subsequent step reviewed. Resource outlay is constrained by the probable success of the method at each step, based on data from the previous one. For example, an extensive profiling network is undertaken only if soundings have shown that the contaminants are present. In some situations, an undefined result may arise whereby the consequent step is a gamble. The balance is then made between the worth of the next step and the gainable benefits, versus the loss of outlain resources in the attempt. For the worst scenario the knowledge of why the method failed is often gained which can be applied to the feasibility assessment at other sites.

5.2 INDUCED CONTAMINATION

5.2.1 INTRODUCTION

The success of salt-water tracing depends on the same factors as for any other DC resistivity use but with additional restrictions due to the tracer properties, and experiment practicality. Thus the required volume, density, concentration, release mode and position of tracer and travel rate all affect experimental results. The contaminant must produce sufficient signal change, and occupy sufficient volume of the surface detection configuration, to be detected (Greenhouse & Harris 1983, Urish 1983, Stollar & Roux 1975). The GSEG Working Party (1988) suggest that the tracer width and thickness should be greater than 20% of its depth, with conductivity 10 times that of the surrounding water. A rapid single tracer pulse of low volume, high concentration, cheaply available electrolyte has been used previously in shallow New Zealand aquifers. A 1000 to 8000 litre, 20% by weight NaCl solution satisfies these criteria and minimises the disturbance to existing groundwater hydraulics.

A number of constraints are assessed to determine whether a natural gradient slug test, monitored by DC resistivity is possible at the Wigram site. Primarily, the response of galvanic resistivity to a salt-water tracer, given the local aquifer geoelectric properties, is examined using a modelling approach devised by Paul White (*pers. comm.* 1990). Curve matching of soundings produces a geoelectric model to which an additional layer, with the properties of a salt water slug is added. The difference between the ideal and the contaminated models is used to determine an optimally sensitive electrode spacing upon which an array network is established around an

injection well. Changes in apparent resistivity readings, in response to the passage of the induced contaminant, are used to characterise the groundwater hydraulic properties.

5.2.2 SOUNDINGS

Three Schlumberger soundings were carried out at Wigram, using an ABEM 300 Terrameter SAS system, with overall range from 0.5 m.Ω-m to 1999 k.Ω-m. The Schlumberger array was chosen for ease of field use and availability of curve modelling software. Orthogonal soundings are undertaken to assess natural variability of the geoelectric log with site orientation. Site factors at Wigram dictated the maximum orthogonal sounding lengths possible to be 80 m and 100 m, as located in Figure 4.7. Sounding 2, oriented as for Sounding 3, encountered surface changes in lithology (from gravelly soil to fine sand) at small MN spacings. The results are therefore excessively noisy at small AB/2 spacings on the sounding plot, and can not be used satisfactorily for curve matching. Hence, only Soundings 1 and 3 are further described.

Field results are initially smoothed using the method of Broadbent (1984) to give apparent resistivity versus AB/2 curves. Preliminary two layer manual curve matching provides estimates of the first and second layer resistivities, and first layer thickness, for an initial model. Refinement thereafter uses the DSIR Groundwater Group SREINS computer program (modified from Merrick 1977), which is based on the iterative least squares optimisation technique of Ghosh (1971). Apparent resistivity values of the initial model are systematically compared to field values, with an iteration loop adjusting the curve to produce a closer fitting model, until the differences fall below a certain root mean square cut off. A number of electrically equivalent, therefore nonunique, geoelectric models for the Wigram site were developed. Percentage error between models and field values were noted, in order to better fit solutions to the field data, however inherent errors of a similar magnitude may result from field method and data smoothing. The most appropriate model was thus primarily selected on the basis of comparison with lithologic logs of wells M35/6102 and M35/6103, with allowance for error minimisation. Sounding curves and corresponding best fit models are given in Figures 5.4 and 5.5 whilst field and model data is given in Appendix 5.1.

Both Soundings 1 and 3 are K or DA (descending anisotropic) type curves,

interpretable in three layers with $\rho 1 < \rho 2 > \rho 3$. The uppermost layer represents a few centimetres of topsoil (33 to 43 Ωm) underlain by unsaturated gravels (1622 to 1846 Ωm) and saturated gravels (294 to 326 Ωm). Depth to the water table at approximately 7 m is more accurately determined in Sounding 1 (6.7 m) than Sounding 3 (9.2 m) but both resistivity estimates are within acceptable bounds given the method constraints.

The Dar Zarrouk parameters, transverse resistance (T) and longitudinal conductance (S), may more realistic describe models (Maillet 1947), and are presented along with the coefficient of anisotropy, λ. Lambda values represent the ratio of maximum to minimum resistivity in the longitudinal versus transverse directions in rock with values generally ranging from 1.1 to 1.3 although higher values are found in clays and thinly bedded sediments. Soundings 1 and 3 at Wigram have λ values of 1.57 and 1.97 respectively.

5.2.3 TRACER DETECTION FEASIBILITY

5.2.3.1 Tracer Targeting

A surface geophysical signal is influenced by earth materials from all depths and lateral distances. In both homogeneous and simple layer models, the signal contribution passes through a maximum before tailing off. Roy & Apparao (1971), following Evjen (1938), defined the zone of this maximum (z_{max}) as a depth of investigation characteristic (DIC). Edwards (1977) disputed the use of z_{max}, preferring instead the use of the median depth (z_{med}), namely *"that depth at which exactly half of the total signal originates from above and one half below"*. The DIC is not equivalent to depth of signal penetration, since deeper layers still contribute to the signal, but optimal results are obtained for selecting an array type and spacing where the DIC is coincident with the target (Roy 1972), in this case the tracer travel depth. (DIC variations for array types used in this investigation are given in Appendix 5.2). Thus, if the tracer travelled at a deeper level than the estimated DIC, a lessened deviation to the baseline geoelectric properties of the site occurs. At some point this deviation becomes unmeasurable. Consequently, it is not uncommon for an introduced slug to "disappear" into an aquifer.

There is a consequently a tradeoff between the optimal array length chosen as having a DIC coincident with the tracer travel depth and the practical array lengths useable at the site. Shorter arrays are easier to deploy and more arrays can be used for the

same amount of total wiring. However, shorter arrays may be blind to the tracer. The maximal practical Wenner array length for use at Wigram is L=40 m, for which the DIC equals 4.48 m (Roy & Apparao 1971) or 6.92 m (Edwards 1977). To achieve the same DIC result for a Schlumberger array, the array length, AB/2, would be between 34.24 and 55.36 m (respective methods as above). At Wigram, the depth to the water table - the nearest surface position at which the salt would have an effect, was at the time of this study approximately 7 m, slightly deeper than the DIC for this array size. Rather than relying on array length selection by this convention, the geoelectric site properties can be utilised to model the detectability of a conductive tracer.

5.2.3.2 Prospective Geoelectric Modelling

Modelling the response of one dimensional geoelectric models to thickness and conductivity changes, is recommended by Mazac *et al.* (1987). In various contaminant studies this approach has been used to characterise the natural scatter of readings due to heterogeneous geological conditions, and to assess detection device sensitivity (Greenhouse & Harris 1983, Greenhouse & Slaine 1986). In this study, a low resistivity layer representing a salt water electrolyte slug (with properties outlined in Appendix 5.3), is added to geoelectric models of Soundings 1 and 3, to ascertain its effect on surface readings of apparent resistivity. All parameters are fixed to prevent the automatic adjustment of the modified model to produce a better fit to the field results. This yields a stand alone independently modelled sounding in which the effect of the tracer (lowering the apparent resistivity readings) is evident. Figures 5.6 and 5.7 show that the effect of the salt is maximised for a Schlumberger array of AB/2 = 40 to 80 m (Sounding 3), or 80 to 160 m (Sounding 1), which would result in resistivity variations of the order of 15 to 80% dependent on the model used. Given that a large number of monitoring points are required for mapping the migrating contaminant, these electrode spacings are too large to be of practical field use. For an array of L=40 m however, the least affected, more conservative contaminant model (1 Ωm, 0.5 m thick, Figure 5.6) still produces a 15 to 30% apparent resistivity change. Thus, for a field experiment with tracer release at 7 m, and a manageable array spacing of L=40 m, a 15 to 30% apparent resistivity decrease is anticipated; however a number of other considerations affect the experiment.

5.2.4 EXPERIMENT CONSTRAINTS

5.2.4.1 Tracer Behaviour

Increasing the NaCl electrolyte concentration in the tracer solution to enable surface DC resistivity detection, also increases the fluid density. Density differences alter flow patterns, the degree of ion exchange on clay particles and organics, and secondary chemical precipitation (which in turn affects aquifer permeability). Whilst in the relatively clay free unconsolidated sediments at Wigram, the latter possibilities are negligible, the former is of concern. The denser fluid may not flow with groundwater, but may sink, or result in density and gravity separation during the test (Grisak 1979). Davis *et al.* (1980) suggest that NaCl concentration should not exceed 3000 mg/l because of density effects, but this may be ameliorated by raising the temperature of the solution. Conversely, immediate dilution of the tracer upon release into the aquifer may occur, negating the need for density correction. Thompson (1980) suggests that dilution by as much as 6 or 7 orders of magnitude may be unavoidable. With either density or dilution effects, the danger is that the tracer is either too dilute, or travelling too deeply, for DC resistivity detection.

Rapid tracer injection may produce radial flow which will initially dominate aquifer hydraulics. This radial effect can be neglected a short distance from the well (Fried 1975) beyond which the existent aquifer flow characteristics dominate.

At the time of the proposed tracer test at Wigram, two wells existed at the site, M35/6102 and M35/6103. The use of conductivity probes installed in either or both of these wells, to independently monitor tracer movement with depth, is the best means of maximising experimental results, given the possibility that surface detection will fail.

5.2.4.2 Tracer Release Position and Volume

It is usual to mix and release a tracer throughout the slotted depth of an injection well, but this requires a substantial tracer quantity. At Wigram the water table was at a depth of 7 m during this investigation, which is several metres deeper than typical surface tracking salt tracer experiments. The injection bore was slotted from 7 m to 36 m with no confirmed extensive impermeable layer bounding the system throughout this depth. Even with a minimal plume thickness of 0.1 m, an optimal salt solution volume of 25 m^3 is required to occupy sufficient array detection space to guarantee the

success of the method (Appendix 5.3). Thus, by extending the release zone, insufficient areal volume is released per unit length, and the bulk of the tracer would lie below the level of the optimal geoelectric signal contributor. Thus, a sectioned off, well defined, tracer release interval is desired.

Continuous injection, large volume, and even recirculating tracer testing may be possible at the Wigram site, but is beyond the constraints of this investigation. Tracer volume is restricted to 4000 litres - the maximum onsite tanker capacity available to the experiment. Salt, at 20% by volume, can pose mixing problems. Continuous fluid circulation, using the tanker discharge pump, and an external recharging pump, whilst slowly adding the salt into the top of the tanker, works well.

5.2.4.3 Array Arrangement

A radial network array arrangement is advantageous for tracer tests when the flow direction is not well defined, because 360° of coverage around the injection point is achieved for relatively few arrays, as compared to a rectangular grid system (Moore 1990). With the use of overlapping Wenner arrays with "a" electrode spacings of 13.33 m (or L=40 m), electrodes are shared, minimising electrode peg numbers and thence cable requirements for the area, compared to an equivalent Schlumberger network. Steel electrodes are wired back to a central electrode configuration switching panel to allow rapid network coverage. In addition, measuring points upgradient of the injection well allow for an assessment of radial flow domination following the initial tracer release. The devised array (Figure 5.8) has the further advantage of superimposing a geoelectric monitoring point over the injection point to allow coincident indirect and direct monitoring methods. The array is symmetrical and elongate parallel to the direction of groundwater flow (indicated by the broad potentiometric survey of the area, section 3.4.3.1), but is sufficiently wide should flow either be slow and widely dispersed, or significantly deviate from this direction. In alluvial aquifers, flow characteristics can vary through a number of scales and local heterogeneities including crosscutting channels can significantly divert flow from the expected direction, as exemplified in Thorpe (1977). Surface tracer tracking can therefore be used as the basis for siting monitoring wells for extended tracer experimentation.

Absolute apparent resistivity readings vary depending on the length, and type, of cable

used in a geophysical survey. Here, the relative (percentage) changes at each monitoring position are recorded so that absolute values and errors arising from array wiring are negated.

5.2.4.4 Water Effects

Surface resistivity measurements are weather dependent. Rain precludes current penetration, since electric charge will travel along the path of least resistance - a wet soil and grass surface. Even small amounts of moisture, e.g. morning dew, can also cause earth leakage along cable lengths. Hence a reasonably lengthy spell of fine weather is required for good results.

Changes in the geoelectric baseline readings are usually due to water content variation and may occur with a fluctuating water table. White (1985) found that pore drainage due to receding rain waters significantly affected a salt-water tracing experiment at Takaka, confusing natural with tracer effects. This can be minimised (and to some extent compensated for), by observing the variation in baseline readings prior to an experiment and at control points outside the tracer flow path during its migration.

5.2.4.5 Summary Assessment

One dimensional computer modelling indicates that apparent resistivity decreases of 15-30% should result from monitoring the transit of an induced salt-water contaminant released just below the water level at Wigram, with a network based on L=40 m Wenner arrays. Other factors have an unknown affect and the experimental planning and method can only try to minimise these. The chosen field set up is illustrated in Figure 5.8.

5.2.5 TRACING EXPERIMENT

Baseline surveys one day, and one hour prior to the experiment showed minimal change (Figure 5.9), with water level change rates insufficient to affect readings.

The injection of 4000 litres of 20% by weight NaCl commenced at 10am Tuesday 26 September 1989 with the release rate into the aquifer of 5.22 l/s. Tracer release was into the 7 to 9 m packed off interval. DC resistivity measurement continued for a period of 192 hours (8 days) with rainstorms inhibiting readings thereafter. Conductivity readings from probes located within the release well, and at various levels

in the observation well were unaffected by weather changes, and continued for a further 20 days, ceasing when values stabilised at background values.

5.2.5.1 Water Levels

Water levels in both the injection and observation well were unchanged during the experiment attesting to the permeability of aquifer sediments since water table mounding was not produced. Water levels were static at 6.87 m below ground level in the injection well throughout the experiment.

5.2.5.2 Resistivity Mapping

The apparent resistivity baseline survey of the Wigram test site prior to salt-water injection is shown on Figure 5.9. The 1% level is taken as a cutoff value to negate measurement and operational errors. There was <1% variation between the two baseline surveys. For the duration of the experiment, apparent resistivity values outside the contaminant path, showed only minor variation, denoting relatively stable background values. Baseline apparent resistivity values range from 670 Ωm to 1690 Ωm, with higher values in the southwest half of the array. Since no topographic variation exists within the array area these differences are attributed to subsurface lithologic variations of porosity, permeability and sediment grain size changes.

Apparent resistivity percentage decrease for various times subsequent to injection are mapped in Figure 5.9. A zone of 3% apparent resistivity decrease developed immediately around the injection well, typical of initial radial flow dominance. With time, the main plume slowly elongates out from the injection well along a 145° orientation with a maximal resistivity decrease just over 4%. The expected 15-30% resistivity changes (section 5.2.3.2) did not occur, which may be a function of tracer sinking or dilution.

At t=108 hours the main plume had spread both eastward and southwestward producing a somewhat convoluted pattern. Throughout this pattern evolution, a separate zone centring at measurement point A, exhibited continued resistivity decreases, up to a 6% maxima. After t=140 hours the main plume had extended sufficiently south to link up with this secondary zone. By t=192 hours, the initial southward (156°) direction of the conductive water body had swung back to the east on an 85° traverse.

In comparing the final (t=192 hour) pattern with that of the baseline resistivities, it is evident that the two high resistivity zones B and C focused on the base resistivities of 1300 Ωm and 1600 Ωm, are central to the resulting configuration. Neither of these zones showed apparent resistivity decreases above 1% except at the final time frame where only one point is affected to this degree. With southward migration, the plumes decreasing effect on the observed readings was more marked at those measuring points at which the baseline resistivities were lower, including the minimum observation of 670 Ωm at Zone A. The absolute resistivity changes constituted a greater percentage of the initial value for these points. Where higher initial resistivity readings exist, the passage of the tracer is consequently masked. Thus at point A the steadily decreasing resistivities are due to the tracer moving progressively southward (145°) having passed undetected through zone B. Likewise transition through zone C was indiscernible. Thus, resistivity decreases at D were noted before the tracer had seemingly entered Zone C. If these higher resistive areas did not exist, a more obvious plume pattern would have emerged from the DC resistivity mapping with the plume slowly progressing southward (145°). Instead a three dimensional inhomogeneous aquifer is reflected.

The mapped geophysical pattern is a two dimensional representation of a three dimensional flow system, in which the solute will move laterally and vertically around flow-impeding zones of lower permeability, porosity, or sediment grain size. Nondetection "holes" in the geophysical map are points at which tracer flow is diverted around or beneath these high resistivity zones, thus being masked (Figure 5.9). Clay is more conductive and less permeable than gravel or sand and the presence of clay lenses may account for this masking effect. The geophysical map thus portrays a myriad of interconnecting flow paths typical of braided river deposits. The patterns also exhibits the inhomogeneous nature of the aquifer, with tracer travel quicker through some sediments than others. Apparent resistivity decrease, with tracer arrival, and increase with migration, at each monitoring point was neither sufficiently marked nor stable enough to employ in groundwater velocity calculations, as illustrated in White (1986, 1988) and Moore (1990).

The observation well lies almost orthogonal to the preferred tracer flow direction. Resistivity decreases were noted here until t=12 hours, with values returning to baseline levels thereafter. The early dominance of radial flow is thus observed prior

to tracer movement with groundwater away from the monitoring well.

5.2.5.3 Conductivity Probes

Conductivity values (corrected following probe calibration) for the injection well are given in Figure 5.10 and for the observation well in Figure 5.11. In the injection well the maximal conductivity recorded is 60.6 mS/cm (0.165 Ωm), equating to a concentration of 42.1 g/l NaCl. This represents a reduction factor of 4.7 times that of the injected concentration (200 g/l), and results from initial groundwater mixing.

In the monitoring well Probe 4, at a depth of 26 m, records conductivity increases prior to the other two probes. Suspecting the deeper passage of the tracer slug, Probe 5 (initially sited at the water table) was repositioned at a depth of 20 m. The repositioned Probe 5 missed the tracer peak, with the maximal value of 5.70 mS/cm (1.75 Ωm, 3.45 g/l) recorded. Probe 4 recorded a sharp elution front peaking at 5.86 mS/cm (1.71 Ωm, 4.06 g/l) in the observation well approximately 2.5 hours following injection. This concentration is a factor of 10.5 times lower than that measured in the injection well, and 49.3 times lower than the solute concentration introduced from the tanker. Probes 2 and 3, at shallower depths of 8.97 m and 13 m respectively, record conductivity values close to background levels, never exceeding 0.6 mS/cm. The main tracer slug is therefore travelling at a depth in excess of 13 m and is diluted just over ten times in the 10 metres between the injection and observation wells. This accounts for the nature of the anomalies arising from the geophysical mapping.

5.2.5.4 Point Dilution Tests

With all three probes (1, 4 and 5), conductivity decline with time is regular and logarithmic as illustrated in Figure 5.11. The release of the conductive solute in the injection well, in which Probe 1 was set, is equivalent to undertaking a point dilution tracer test as outlined in Appendix 4.3. If travel to Probes 4 and 5 in the observation bore is neglected, the solute peak is analogous to a rapid tracer introduction. The tracer conductivity decline therefore represents a point dilution experiment where solute dilution is a function of groundwater flow. Point dilution tests for Probes 1, 4 and 5, give groundwater average linear velocities of 8.52, 3.58, and 6.34 m/day respectively (Appendix 5.3).

5.2.6 RETROSPECTIVE GEOELECTRIC MODELLING

Modelling the effects of a conductive slug on the original geoelectric model of Wigram site (Figures 5.6 and 5.7) was re-evaluated in light of field observations. Rather than use the induced tracer concentration of 200 g/l, the maximum concentration observed at the observation bore is used. An open ended base layer with a resistivity of 38 Ωm (derived in Appendix 5.3) represents a more fitting geoelectric model than a single layer as in the pre-experimental modelling, and accounts for initial dilution and sinking effects. A range of fixed models, with the top of the salt layer varying at 1 m depth intervals from 7 m to 30 m was treated by the SREINS programme. The percentage difference between the apparent resistivity generated for each model at AB/2 = 20 m (synthesising a Wenner array spacing of L=40 m) and the original ideal model derived from curve matching of field results was calculated as outlined in Appendix 5.3. As illustrated in Figure 5.12 the percentage change in resistivity declines logarithmically with the increase in depth to the salt tracer.

During field experimentation the migrating tracer was observed at depths of 20 m and 26 m in the observation well, but not by probes at a depth of 14 m or shallower. The expected resistivity decrease for the slug if it flowed below 14 metres is just over 18% (Figure 5.12), halved to 9% at a flow depth of 20 metres, and just below 5% at a depth of 26 m. The latter two values are of the order of change (maxima 6%) observed by geophysical mapping. The tracer therefore sank quickly regardless of being released at a shallow constrained position in the borehole.

5.2.7 DISCUSSION

Induced contaminant tracing at Wigram was used to establish groundwater flow parameters for modelling. This was of secondary concern to the main requirements of assessing leachate behaviour at the Paparua County Council Landfill, so the experiment was not repeated, but the following observations are noted for others attempting similar experiments.

1. The undertaking of tracer tests is a continual gamble, since aquifer behaviour and geophysical response are never ideal. However the use of multiple techniques and a systematic approach optimises the possibility of success. One dimensional geoelectric modelling gives a more informed understanding of tracer effects than relying on conventional array selection by depth of investigation characteristics. Post

experimental modelling successfully synthesises the behaviour of the tracer slug and may explain, as in this investigation, why the resistivity changes suggested in pre-experimental models are not achieved. Pre-experimental modelling upon which the decision to proceed with the resistivity technique was based, used an apparent resistivity layer based on the injected tracer concentration. The dilution of this tracer upon encountering aquifer water was not assessed, nor could its full effect be known unless the salt tracer test was re-enacted. The utility of post-experiment modelling is therefore twofold:

i. to clarify hydrogeochemical results following field tests to explain the discrepancy between predicted and actual field results; and

ii. to incorporate this information in further tests planned for the site to select optimal solute concentration and volume, and the most effective surface resistivity profile arrangements.

2. An initial mapping experiment should be used to site monitoring wells that could be incorporated in ongoing experiments. The combination of surface geophysics with downhole conductivity probes yields a more complete explanation of tracer behaviour (including sinking and dilution effects) and is recommended for tracer experiments.

3. Groundwater geophysical observations would have been improved with regard to depth effects with either of the following additions. Repeated soundings at varying positions downgradient of the injection well, but beyond the zone of initial radial flow, would reflect the actual degree of electrolyte effect rather than that assumed by geophysical modelling. The amount of sounding variation with distance from the injection point could thus be used to interpret sinking effects. In addition, better utilisation of the array network is possible by incorporating measurements of the longer array lengths (i.e., every second electrode). A stacked array is therefore intermediate between a single array network and a series of soundings. This would then constitute a separate monitoring network penetrating three times deeper than the original, with minimal additional effort. This was undertaken at the Wigram site but only late in the field experiment; without base values for comparison, resistivity changes at successive monitoring times could not be interpreted. However, more rigorous planning of stacked arrays could form the basis of pseudosection, as well as mapped, tracer monitoring, enabling three dimensional contaminant detection as in

Russell & Higer (1988).

4. Electrode configurations can be more suited to sounding or profiling as outlined in Appendix 5.2. However array orientation can also have a marked effect on tracer detection. In recent work by White (1991), six array configurations were compared on the bed of the Rakaia River, Canterbury. In response to the same tracer experiment, an α Wenner array detected up to 28% resistivity changes, whilst a resistivity rectangle array detected 60% anomalies. In addition Wenner arrays aligned with the direction of groundwater flow were found to have a greater sensitivity to the passage of salt water than equivalent arrays orthogonal to flow. This validates the initial selection of a radial array network about an injection point in tracer tests, but suggests with experiment repetition that a box network of arrays all parallel to the flow direction be used for optimum detection results.

5. In the Wigram experiment water level monitoring probes consistently clogged with reprecipitating salt following tracer injection, so for rapid measurement of level changes a downhole pressure gauge system is recommended.

6. Although this experiment shows that surface mapping by DC resistivity can be used where depth to water table is up to 7 m, it is more satisfactorily confined to shallower conditions where physical access to water sampling can verify results.

5.3 EXISTENT CONTAMINATION

5.3.1 PREVIOUS WORK

A variety of existent contaminants, whose presence and migration have been studied by geophysical techniques, include salt water intrusion (Swartz 1937, Van Dan & Meulenkamp 1967, Bates & Davidge 1983), and migration of: hot mineralised water in fissure zones (Hatherton *et al.* 1966), acid mine drainage (Merkel 1972, Ebraheem *et al.* 1990), oil (Mazac *et al.* 1989), liquid waste from uranium bearing cold scrap evaporation ponds (Urish 1977 in Mazac *et al.* 1989), sulfite liquor (Hackbarth 1971), oil field brine from holding ponds (Fetter 1984 after Reed *et al.* 1981), fellmongery ponds and effluent irrigation from freezing works (Moore 1990), and chemical residues at Love Canal (Benson 1989). The application of geophysical methods to the study of landfill sites, and leachate migration, is a recent development, but is becoming commonplace. A variety of methods have been used.

Site characterisation using seismic methods was undertaken by Calkin (1989) to detail the bedrock structure beneath a landfill. Rodrigues (1987) combined seismic with gravity methods to examine a landfill suspected of contaminating a town water supply with trichloroethylene, with the results used to site monitoring wells.

Where used to detect leachate, geophysical studies can be proactive or retroactive. For proposed landfill sites, soundings and profiles undertaken prior to landfilling provide baseline surveys with which later post-filling readings are compared. This proactive approach eliminates the effects of lateral variation in geologic materials simplifying the interpretation of geophysical results, and gives advance warning of leachate development and migration prior to it becoming problematic (Klefstad 1975, Greenhouse & Harris 1983). This "prevention is better than cure" philosophy is outlined by Greenhouse and Monier-Williams (1985) for a hypothetical landfill, in which the possibility was recognised that no conclusive results from geophysical patterns would emerge for years. Regardless, the emphasis is on the benefit of preparedness that proactive geophysics offers. A proactive case study, involving galvanic DC resistivity soundings and EM to augment a monitoring well programme, is outlined by Rumbaugh *et al.* (1987) for the Oaks Landfill, Maryland. Bimonthly monitoring of 282 EM stations positioned along the 14,100 ft (4700 m) long landfill perimeter was used to establish seasonal changes in baseline surveys.

The retroactive approach in response to suspected leachate presence or migration is well established. Galvanic DC resistivity methods outlined in Urish (1983) and Benson (1989) are most commonly used in case studies typically incorporating Schlumberger or Wenner soundings, and Wenner profiling (Cartwright & McComas 1968, Seitz *et al.* 1972, Stollar & Roux 1975, Klefstad *et al.* 1975, Rogers & Kean 1980, Russell & Higer 1988, Carpenter *et al.* 1990). Increasingly the trend is toward the use and comparison / contrast of multiple techniques to optimise data collection, as the success of each method is site dependent.

This view is endorsed by Stierman (1984) who incorporated spontaneous potential, DC soundings, dipole-dipole profiles, and miss-a-la-masse methods at the Stringfellow site, California. VLF (very low frequency electrical technique) was used jointly with EM and galvanic DC resistivity mapping at the Borden site in Ontario by Greenhouse & Harris (1983). Maps of the contaminant plume produced using each method were

similar but DC suffered from natural scatter due to lithologic variations, and EM was consequently recommended for its lower cost and higher resolution. Stewart & Bretnall (1986) validated the use of VLF in their comparison between these three methods. Sweeney (1984) used both dipole-dipole, and EM mapping and monitoring, at a landfill in the California Coastal Ranges, where the methods complemented each other. The EM and DC combination was used by Greenhouse & Slaine (1986), and in the study of two industrial locations and a landfill by Stierman & Ruedisili (1988). In the latter study, DC was less susceptible to cultural noise but EM was quicker and cheaper to use. Neither method, as with any geophysical technique, is successful if there is significant natural or cultural variability.

All the studies mentioned above were undertaken to detect the presence and/or migration of leachate. Carpenter *et al.* (1990) is particularly notable as it is the first attempt to study the geophysical response of internal landfill stratigraphy using Wenner and Schlumberger soundings. The extent and thickness of refuse, cover material and leachate was assessed with reasonable accuracy at the Mallard North Landfill in DuPage County, Illinois.

Within New Zealand, examples of surface geophysical techniques used to delineate contaminant patterns are sparse, and are restricted to galvanic resistivity. Risk (1980) applied the technique to trace wood processing effluent at Kinleith. White (1985, 1986) undertook DC resistivity studies at the proposed Takaka sewage pond, Nelson, and the Renwick Refuse Tip near Blenheim, to evaluate groundwater flow prior to the installation of each facility. These are not really proactive examples in that changes to baseline values were not sought to assess leachate development with time, but they serve as good examples of induced contaminant tracking. Moore (1990) includes an example of both induced and existent techniques, with a DC resistivity Wenner grid used to map a contaminant plume emitted from a Freezing Works Fellmongery Pond at Waingawa in the Wairarapa. A 300 m long plume was successfully detected, with hydrochemical sampling indicating its extent at 800 m.

Broadbent (1991) is the first published example of surface geophysical methods being used to attempt leachate plume delineation at a landfill in New Zealand. A low resistivity zone, attributed to leachate plume emanation from a completed sector of the Kaiapoi Landfill (20 km north of Christchurch), was detected on the basis of nine

galvanic resistivity profile transects. Similar investigations at the Paparua County Council Landfill have been enhanced by geophysical landfill stratification and the incorporation of hydrochemical sampling to assist the interpretation of geophysical results.

5.3.2 METHOD

The stepwise approach outlined in Section 5.1.3 is used to characterise the Paparua County Council Landfill. Six DC resistivity soundings, carried out at three locations at the P.C.C. Landfill, are shown on Figure 2.7. Locations DOG and BORE are sited on areas known to have been infilled with refuse, and SHEEP is located downgradient of the extraction limit. To provide information in excess of that available from a single sounding method, both Schlumberger and Wenner arrays (with α, β, and γ variants) were undertaken at each location, named hereafter by a location and method label, e.g., DOG.W. Of the arrays used, the α Wenner has the best vertical resolution (Appendix 5.1) and is applicable for SREINS computer modelling. Although the curve modelling facility was unavailable for β and γ arrays, results can immediately highlight lateral variation and instrument error effects on a sounding plot, as outlined in Carpenter & Habberjam (1956), Habberjam & Watkins (1965), Barker (1979), Ackworth & Griffiths (1985). Characterisation of the underlying lithologies as well as the landfill structure was required in order to contrast tipped versus untipped areas, and to identify leachate zones. Sounding analyses were undertaken as in Section 5.2.2, but with reference to wells M36/4017, M36/4018, M36/4120, and M36/4302.

A single-channel seismic refraction traverse was also undertaken at the same location as for the SHEEP sounding (Figure 2.7).

5.3.3 SOUNDING RESULTS

5.3.3.1 DOG and BORE

For the two refuse infilled locations, DOG and BORE, both Wenner and Schlumberger soundings follow H type curves (Hummel type with minimum) with prominent minimas of 10 to 12 Ωm (Figures 5.13 and 5.14), suggesting a three layer model ($\rho 1 > \rho 2 < \rho 3$) of soil atop refuse, in turn overlying gravel. However, a flattened extended curve minimum particularly in DOG.S and DOG.W, suggests the possibility of additional layers. The undulations on this curve section are consistent for all the Wenner array variants suggesting additional layers as in a QH model ($\rho 1 > \rho 2 > \rho 3 < \rho 4$)

with Layer 1 soil cover, Layer 2 unsaturated refuse, Layer 3 saturated refuse, and Layer 4 the underlying alluvium.

Curve matching has produced models closely matched to lithologic logs. The tops of the saturated refuse indicated on Schlumberger resistivity models at DOG (6.3 m) and BORE (5.2 m) are within 0.3 m of water levels recorded in monitoring wells installed close to each location. The base of this saturated refuse layer was similarly close to the refuse emplacement level known from site history and lithologic logging at BORE.S. The water table is approximately 3 m higher than is represented on DOG.S.

A saturated refuse layer with a resistivity of 5.1 to 6.7 Ωm is defined from Schlumberger models and 2.3 Ωm from DOG.W which compare favourably to leachate resistivities, as low as 6.25 Ωm (160 mS/m), measured in monitoring wells at the DOG site (Chapter 6). Carpenter *et al.* (1990) found similar values of 2 to 7 Ωm for saturated refuse, whilst general unsaturated refuse values of 9 to 19 Ωm are comparable to those found in this investigation (19.6 to 30.1 Ωm). BORE.W was best fitted with a model which did not include a leachate layer but the curve is less easily modelled than the smoother equivalent Schlumbeger sounding. Underlying saturated gravels at the P.C.C. Landfill with resistivities of 303 to 565 Ωm, contrast markedly with the low resistivity overlying layers. The surface patchy soil cover varies in thickness and resistivity as reflected in the uppermost model layers.

BORE.S was best modelled by five layers ($\rho 1 > \rho 2 > \rho 3 < \rho 4 < \rho 5$) with an additional 50 Ωm layer representing silty sand immediately underlying and confining leachate, as noted in lithologic logging of wells. At DOG, refuse was immediately underlain by 2 m of gravel before silty sand was encountered. The deepest interpreted model layer does not extend deep enough to cover the introduction of a fines unit. The second layer on the DOG.S model is taken to be an mixed soil and refuse landfill cap with only thin topsoil represented by the first layer.

At large "a" spacings, electrodes at the northern end of the DOG sounding were outside the landfill site; thus resistivity readings are contributed to by both refuse infilled and virgin areas. These lateral effects are reflected by the erratic undulations of the DOG.W sounding curves at larger electrode spacings.

5.3.3.2 SHEEP

Best fit models for the SHEEP location, downgradient of the refuse infilled section of the landfill, are illustrated in Figure 5.15, with field data and the interpreted model given in Appendix 5.4. Since the site lay outside refuse deposition limits, both SHEEP soundings are H type curves, but the minimum is less (around 100 Ωm) than that observed at DOG or BORE. The model, consistent for all Wenner array variant sounding curves, reflects unsaturated surface gravels and sands with very high apparent resistivities. The top of Layer 2 at 2.2 m deep, may reflect the position of the water table (recorded at 2.57 m deep) or the top of a silty sand unit. This latter possibility is favoured, as the layer resistivity (77 Ωm) is similar to that of Layer 4 in BORE.S (50 Ωm), lower than typical saturated gravel values (303 to 565 Ωm), and higher than the resistivities detected at BORE and DOG for leachate layers (2.3 to 6.7 Ωm). Driven monitoring wells ended in silty sand at depths between 5.5 and 8.5 m at the SHEEP locality, but the top and base positions of the unit are unknown. Resistivity soundings indicate some change, probably to saturated gravels, at a subsurface depth between 6.8 and 9 m.

SHEEP sounding curves are similar to those recorded at the Kaiapoi Landfill (Broadbent 1991) wherein layers with resistivities <100 Ωm but lying within 15 m of the ground surface are attributed either to fine lagoonal sediments or conductive leachate. Both sites, and by extension any site in Canterbury suspected of leachate contamination in alluvium, present the possibility of layer suppression, with a low resistivity layer sandwiched between higher resistivity layers. With a tendency for most current to flow through this low resistivity zone, additional middle layers (thin compared to their depth) have minimal effect on the resistivity curve and can therefore go undetected. Thus, both leachate and fine sediment may be present but are indistinguishable due to layer suppression, and clarification by groundwater sampling is required. At Kaiapoi only one sounding appeared to be influenced by leachate on the basis of groundwater conductivity values. Conductivity probe readings in well M36/4315 near SHEEP, gave a groundwater resistivity value of 34 Ωm (29 mS/m), similar to values cited by Broadbent (1991) for uncontaminated wells.

5.3.3.3 Sounding Assessment

With more points defined per decade than equivalent Wenner soundings, Schlumberger soundings produce smoother curves, less affected by lateral variation at

small electrode spacings and are advocated by Carpenter *et al.* (1990) for landfill stratification characterisation. Although the Wenner technique worked well at the SHEEP locality, curve modelling experience for the other locations at the P.C.C. Landfill endorses this recommendation.

Landfill stratification is clearly resolvable using galvanic resistivity soundings. Curve matching clearly distinguishes leachate from overlying unsaturated refuse and underlying alluvium, as exemplified by Carpenter *et al.* (1990) and endorsed by this investigation. Both BORE and DOG soundings exhibit low resistivity zones typical of contaminated groundwater. At SHEEP, soundings are devoid of such low resistivity effects, but do exhibit a lower resistivity middle layer attributable to either fine sediment, diluted leachate or (with suppression effects) the existence of both these features. Invasive techniques, lithologic logs and water sampling indicate this unit to be in situ uncontaminated silty sand. Thus, while the extension of soundings to profiling should indicate contaminated zones of leachate presence, two points are noted:

1. leachate may be indistinguishable from fine underlying sediment, and is unambiguously detected solely on the basis of geophysics only where apparent resistivity values are less than 70 Ωm; and
2. groundwater sampling is recommended to enhance and clarify geophysical findings.

5.3.4 PROFILING

The existence of a low resistivity layer representing a conductive leachate zone was observed from resistivity soundings at the P.C.C. Landfill. Geoelectric mapping of the site, may detect areas of preferred leachate development and/or migration by targeting this leachate layer.

Two superimposed linear Wenner arrays using common electrodes were chosen to enable the simultaneous measurement of two depth profiles. Pseudosections could thus be constructed, with minimal additional field work compared to a single layer array, to assist in distinguishing the effects of shallow geological structures from deeper ones, as recommended in Zohdy (1974) and noted in Section 5.2.8. Layer 1 profiles are based on Wenner arrays with "a" spacings of 15 m (Layer 1) which is equivalent to a depth of investigation of 4.95 m (Roy & Apparao 1971) to 7.78 (Edwards 1977).

Layer 2, with a=30 m has an investigation depth of 9.9 m to 15.56 m. Thus the geoelectric properties of the aquifer just below the water table (4.5 to 5 m), and within the sediments underlying the landfill, are measured.

The applicability of the technique was assessed using three trial profile traverses (AA', BB', CC'), located in Figure 5.17, which encounter both refuse infilled, and virgin land. Electrode positions were located by numbered wooden pegs, easily seen but minimally affected by stock, or by farm and public vehicles. Peg positions were surveyed using EDM surveying equipment.

Geophysical profiling in October 1990 was undertaken by two persons over eight hours using a system that minimised movement of four cables, steel electrodes, and field personnel, but which allowed rapid alternate Layer 1 and 2 readings to be taken. Where either barbed wire or aluminium clad fences were traversed, care was taken to prevent entanglement. Long unhindered profile lengths were preferred for rapidity of the field system employed.

The success of the trial profiles necessitated wider mapping. A profiling grid (Figure 5.17) was designed to adequately map the site, using long profile lengths, but not to oversaturate readings and therefore effort, by using too dense a coverage (Urish 1985). Further grid extension would depend on profiling results from this field arrangement. The grid took approximately three days to survey and peg out, and survey details were collated to produce a topographic map of the locality.

Two persons traversed the 325 peg grid, and recorded the 251 Layer 1 and 222 Layer 2 readings over 21 hours, in November 1990. Profile NN' in the southwest along the old river bank was repeated several times. Electrode contact problems in the dry riverbank sands caused erroneous fluctuating readings which were remedied by dousing the contact points with water to improve electrode contact (Benson 1989).

5.3.5 RESULTS

5.3.5.1 Trial Profile Lines

Apparent resistivity readings from the field trial lines of Layers 1 and 2 sited are given in pseudosections in Figure 5.16. A logarithmic scale to divide apparent resistivity readings into contourable units is used (after Greenhouse & Harris 1983). This

prevents contours from clustering around a contaminant source affording more detail in low-response areas (Greenhouse & Slaine 1986). For Layer 1 (a=15 m), low apparent resistivity values (down to 16 Ωm) are found in the centre of profile CC' and along the northern end of AA'. Values increase southward to the middle of AA' (maximum 234 Ωm), then decrease south of the AA' BB' intersection to end again at higher values (144 Ωm). Layer 2 similarly shows a pocket of lower resistivity values (down to 38 Ωm) along CC' and to the north of AA' (45 Ωm), increasing southward.

Pseudosections (Figure 5.16), illustrate the increasing resistivity with depth. Contours flatten to produce a tabular shaped pocket of lower apparent resistivity values (hence higher electrically conductive material) near the surface. These patterns correlate well with known extraction and dumping history from aerial photographs of the site, with the deepest extraction area in the northeast corner. Most of the area bounded by Carrs and McTeigues Roads is extracted and refuse infilled land, except for the southern end of line AA'. Aerial photographs show an unextracted block in the middle of AA' (Figure 2.7) projecting to the west which correlates with higher resistivities at this position. Higher apparent resistivity values in these locations represent virgin saturated gravels and sands. Layer 1, near surface, low resistivity values coincide with positions of refuse emplacement and exhibit resistivities representative of saturated refuse. Layer 2 values representing depths below the known extraction level indicate the existence of leachate below the refuse field. This conductive zone appears constrained with depth (note the flattened contours), which is related to either rapid leachate dilution, or to some lithological control, possibly the existence of a lower bounding silty sand. Spurious results due to the existence of cultural obstacles (e.g., wireline fences and aluminium clad fences around the Greyhound Track, and overhead transmission cables) are not observed in the results.

5.3.5.2 Profile Grid

Figures 5.17 and 5.18 present mapped geoelectric values obtained from field resistivity profiling. Apparent resistivity values for Layer 1 profiles clearly mimic the known extraction and refuse deposition areas of the site with low apparent resistivity values centring about the middle to eastern end of the Greyhound Track, then slowly increasing to the SSE but rapidly increasing in value toward each road boundary. The lowest values (down to 14.8 Ωm) cluster in the northeast extracted corner, and are attributed to the presence of leachate. Higher apparent resistivity values are found

to the southwest underlying the Kart Track, and to the east of Carrs Road and north of McTeigues Road where there are virgin saturated gravels and sands.

Layer 2 shows a similar pattern but is not as constrained by road boundaries, as illustrated by the 138 Ωm contour to the east of Carrs Road. The lowest resistivity values (down to 23.6 Ωm) are also centred about the middle east of the Greyhound Track. The 76 Ωm contour divides this conductive zone into two areas, with the smaller patch found to the southeast of the fenceline. The 103 Ωm contour similarly indicates a small separate body to the southeast, but in general the pattern of this contour remains the same in Layer 2 as in Layer 1. Pseudosections based on combined Layer 1 and 2 data clearly show the resistivity increases with depth. A slight eastward extension of the lower resistivity zone (based on the 76 Ωm contour) in the deeper profile is indicated and is consistent with the expected leachate migration direction defined from potentiometric surveys.

Elevation changes are slight compared to DC resistivity profile length and are not thought to affect apparent resistivity readings, as they are homogenised over the profile length. The topographic map shows the mounded area where refuse was deposited and subsequently capped above the original road level, as indicated from aerial photographs. Mounding is most prominent in the area of the Greyhound Track. Apparent resistivity patterns mimic topography, but the correlation is probably mapping refuse positions. Geophysically detectable leachate has therefore not migrated significantly from its formation position. Similarly, the flattening out and rapid increase of apparent resistivity contours with depth beneath the refuse emplacement level may result from either rapid leachate dilution or confinement by the silty sand aquitard underlying the landfill. Further investigations to distinguish these possibilities are required.

Greenhouse & Harris (1983) acknowledge that fine sediments below the Borden Landfill could have produced anomaly patterns irrespective of aquifer contamination. This point is equally applicable to the Kaiapoi Landfill (Broadbent 1991) and the P.C.C. Landfill. However, at the P.C.C. Landfill, the underlying silty sand is found throughout the profiled area not merely in low resistivity locations and the resistivity values are far too low to merely reflect sediment geoelectric properties.

The possibility of conductive groundwater contaminants originating from other pits or tips or industry to the north of the McTeigues Road landfill boundary and moving into the study area is acknowledged, but is not indicated by resistivity profiling.

5.3.6 SEISMIC REFRACTION

The single-channel, 105 m long, seismic traverse was undertaken at the SHEEP locality using MD-9A BISON SEISMOGRAPH. Two units were delineated: soil and dry sediment with a seismic velocity of 370 to 522 m/s, and an underlying undisturbed but saturated sediment of 1498 to 1840 m/s velocity. Although the interface between the units was relatively uniform, at a depth of 6.27 to 7.8 m, it does not correlate well with the water table depth, known from monitoring wells to lie 2.6 to 3 m below the surface. This insensitivity of the seismic technique to the water table is common in Canterbury Plains alluvium (Broadbent 1978). The level indicated by the seismic velocity change is similar to a lithologic contact indicated by DC resistivity soundings at SHEEP, of 6.8 to 9 m, and probably reflects the contact between silty sand and underlying gravels encountered to a depth of 8.5 m in monitoring wells.

Following the first 10 m of the second leg of the traverse, which gave velocity values of the surface sediment, difficulty was experienced in picking the first arrival of the signal response, and the corrected plots (Figure 5.19) do not exhibit a critically refracted ray. This is characteristic of a deeper layer exhibiting a lower velocity than the one overlying it (Parasnis 1979), and may be due to inhomogeneous uncompacted refuse beneath a higher velocity capping soil. Carpenter *et al.* (1991) found a similar effect on seismic traverses over the Mallard North Landfill.

In the Mallard North study, the top of refuse was not picked by critically refracted head waves on any of the 30 seismic lines undertaken. Wave attenuation was attributed not to low velocity refuse, but to both severe attenuation of waves in the cover preventing energy penetration into refuse, and velocity inversion, with a clay layer lying within refuse. Such clay interlayers were not detected in bores at the P.C.C. Landfill. However, clay interlayers would be common to many landfilling sites due to the practice of periodically covering over waste cells.

Whilst seismic depth characteristics are not discernable from traverses across the top

of landfills, it may be possible to use seismic refraction to delineate landfill boundaries where the transition between rock and refuse below a surface cover is sharply defined, as typified by an extracted and refuse-infilled pit or quarry.

5.4 SUMMARY

Surface geophysical (particularly geoelectrical) techniques are well established for proactive landfill site characterisation, or retroactive groundwater contaminant detection in either existent or induced contaminant situations where cultural/topographic obstacles, or lateral changes in lithology, do not significantly affect the geophysical signal. DC resistivity was therefore chosen for contaminant detection at two experimental sites: the Wigram RNZAF Base, and the Paparua County Council Landfill.

Orthogonal DC resistivity soundings at the Wigram RNZAF base were modelled with a thin topsoil layer (33 to 43 Ωm), an unsaturated gravel layer (1622 to 1846 Ωm), and basal saturated gravels (294 to 326 Ωm) with the interface between the two latter units correlating well with the depth to the water table established from bores at approximately 7 m. Adding a conductive layer to the sounding models synthesised the effect of a 20% by weight salt-water tracer slug on apparent resistivity values. Decreases of the order of 15 to 30% were indicated if a practical field Wenner array spacing of L=40 m was used for tracer detection.

A rapid, low volume (4000 l), natural gradient salt-water tracer test was undertaken. Tracer movement was monitored using a surface resistivity profiling network for a period of 8 days, and by downhole conductivity probes for a further 20 days. Background apparent resistivity values (ranging from 670 to 1690 Ωm) were stable throughout the experiment. The initial radial flow of the tracer was undetected by water level observations, but was indicated by the immediate formation of a zone of 3% apparent resistivity decrease centred on the injection well.

Contaminant monitoring confirmed ground water flow to be oriented 145°, but the mapped pattern was complicated by "non-detection holes" where it is thought the tracer moved around and beneath less permeable zones. The tracer pattern thus reflects three dimensional inhomogeneities typical of river deposited aquifer sediments, but could not be used to calculate tracer velocity or dispersion characteristics.

Experimental concerns including tracer "disappearance", due to sinking beyond the resistivity detection range and to density and dilution effects, were confirmed. Sharp tracer elution fronts detected by conductivity probes indicated that the tracer was diluted by a factor of 4.7 by initial mixing into groundwater, and by a factor of 10.5 in the 10 metres between the injection and observation wells. The tracer migrated at a depth in excess of 13 m with a velocity of 3.58 to 8.52 m/day.

Retrospective computer modelling, using the diluted tracer conductivity observed by probes in the injection well, indicated initial apparent resistivity profile values would decrease by 9%. Given the assumptions involved, this is consistent with the 6% decrease observed in the experiment. The use of conductivity probes and retrospective geoelectric modelling is therefore endorsed in the planning of subsequent tracer tests, as it significantly increases the understanding of tracer behaviour. In addition, "stacked" detection arrays and soundings aligned with the groundwater flow direction may significantly enhance experimental results by maximising tracer detection, and are recommended.

Two Schlumberger and two (α, β and γ) Wenner soundings effectively delineated a four layer landfill stratigraphy at the P.C.C. Landfill, with soil and unsaturated sediment values of 56 to 307 Ωm, unsaturated refuse of 19.6 to 30.1 Ωm, saturated refuse/leachate of 2.3 to 6.7 Ωm, and deeper saturated gravels of 303 to 565 Ωm. At one location, an additional 50 Ωm layer, representative of silty sand found in lithologic well logs, was incorporated in the sounding model above the gravels.

The possible presence of a leachate layer, whose effects were suppressed in Wenner and Schlumberger soundings undertaken outside the refuse deposition boundary, was negated by conductivity probe values of groundwater which showed the location to be uncontaminated. Rather, soil (3655 to 5710 Ωm) is underlain by silty sand (77 Ωm) and saturated gravels (212 to 238 Ωm).

The success of "trial" profile lines led to more extensive profiling. "Stacked" Wenner profiles, with "a" spacings of 15 and 30 m, attest to leachate confinement both laterally, within the boundaries of the P.C.C. Landfill, and with depth. Values increase rapidly in these directions, particularly at the north and eastern landfill boundaries where undisturbed and uncontaminated sands and gravels are indicated by apparent resistivity

values up to several hundred ohm-m. Apparent resistivity values representative of leachate (as low as 14.8 Ωm) are centred about the northeast most deeply extracted and refuse infilled sector of the site, with some migration in the direction of groundwater flow to the southeast as indicated by the deeper penetrating profile network.

There is no evidence of contaminants moving into the site from upgradient sources, whilst cultural obstacles, elevation changes, and the underlying silty sand, do not unduly influence apparent resistivity readings.

At an undisturbed location within the P.C.C. Landfill boundary, seismic velocities were established for surface sediment and soil of 370 to 522 m/s and the underlying saturated sediment of 1498 to 1840 m/s. The 6.2 to 7.8 m deep contact between the units is unrelated to the water table, but may reflect a lithology change from silty sand to gravel. Traversing from virgin land across the refuse infilled gravel extraction boundary of the Landfill, the seismic response ceased to exhibit a critically refracted ray. This is characteristic of high velocity material overlying low velocity material, and may be due to refuse underlying soil. Consequently, whilst depth information is unavailable from this segment of the seismic traverse, the technique may find wider service in the detection of laterally defined landfill boundaries.

CHAPTER 6 HYDROGEOCHEMISTRY

CHAPTER 6 HYDROGEOCHEMISTRY

6.1 INTRODUCTION

Geophysical and geological data indicate both the presence and spatial distribution of leachate at the Paparua County Council Landfill. The site is therefore of immediate environmental concern with regard to groundwater quality degradation. However, the full environmental status of the site can only be determined by characterising the chemical nature of the leachate and the internal landfill processes controlling leachate constituent production and transport. In this way both the health consequences and a temporal understanding of the site behaviour are achieved.

Hydrochemical sampling was undertaken to assess the impact of leachate on groundwater which is abstracted from shallow wells in the Wigram / Halswell locality for use as untreated drinking and irrigation water. The New Zealand Drinking Water Standards, referred to in this test as the Guidelines, are therefore used as a reference for comparison between samples. "Contaminated water" is used to describe water samples in which any chemical constituent exceeds the Guideline level, as opposed to the *sensu stricto* definition outlined in Chapter 3.

6.2 DECOMPOSITION PROCESSES

The following outline characterising landfill decomposition is based on Pohland (1980, 1986), Pohland *et al.* (1983), Harris & Lowe (1984), Knight (1984), Baedecker and Back (1979), Robinson (1986), Lema *et al.* (1988), and Brown (1978), and relates to Figure 6.1. For further detail of specific reactions and microbial behaviour in landfills, the reader is referred to Lu *et al.* (1985).

Decomposition within a landfill is related to the action of microorganisms which break organic waste down into stable compounds, initially under aerobic conditions; then with oxygen use and non-replacement, anaerobic conditions ensue. Five distinct decomposition phases (Figure 6.1) are recognised in this biological reactor. The initial biological and chemical reactions, and the longevity of each phase, are governed by climatic and site factors including the initial waste composition, as outlined in Chapters 1 and 2. Thereafter, the partial or total anaerobic degradation of the waste produces intermediate products, which affect the type and rate of ongoing biological and chemical processes. With time, most of the hydrolysable organic matter becomes

fermented, and the reaction products are released as solutes / solvents in the leachate load. As older waste is decomposed, fresher more easily degraded waste is attacked. As time passes this fresher material becomes a smaller fraction of the landfill. The age of the site can therefore be related to the processes acting on the waste as indicated by gas composition and various chemical indicators, including conductivity and pH changes in the leachate (Lu *et al.* 1985). However, Pohland (1986) noted:

> *"No landfill has a single 'age', but rather a family of different ages associated with the various sections or cells within the landfill complex and their respective progress toward stabilization".*

Thus the site age and the refuse age in any one landfill section can significantly differ with a number of stability phases found at any one site. Stabilization is controlled by varying physical, chemical and biological properties responding to the heterogeneous nature of the waste and the setting of each specific section. These properties include the availability of oxygen and moisture, temperature, presence of microbial inhibitors, and type and behaviour of microorganisms. The characteristics of each stabilisation phase are nevertheless distinct, as outlined in the sections following.

6.2.1 PHASE I - INITIAL

Except in very dry or cold localities, a specific microbial community is established in the refuse immediately following its burial. The proliferation of microbes may follow a lag period controlled by the moisture content of the waste or its accumulation from precipitation percolating into the waste pile. Once established, the organisms begin to feed on proteins, carbohydrates and lipids in the organic matter.

6.2.2 PHASE II - AEROBIC: TRANSITION

This stage is characterised by the bountiful growth of aerobic microorganisms which convert existing oxygen to carbon dioxide and water, accompanied by heat, up to 60^0C (Farquhar & Rovers 1972 in Lu *et al.* 1985), as follows:

$$\underset{\text{organics in solid waste}}{6\,(CH_2O)X} + 5\,O_2 \xrightarrow{\text{aerobes}} CH_2O + 5\,CO_2 + 5\,H_2O + 6\,X + \text{ENERGY}$$

where X includes ligands such as PO_4^{3-}, SO_4^{2-}, NO_3^{2-}, NH_3.

The organic products of this process include (water insoluble) volatile fatty acids, amino acids, carbohydrates and sugars in measurable quantities. The heat produced can inhibit bacterial growth and survival, and may obstruct the onset of Phase III.

Ferrous and other metals are initially oxidised; however primary electron acceptors shift from oxygen to nitrates and sulphates as CO_2 displaces O_2 in the landfill gases, and reducing conditions (with decreased Eh) are established. Sufficient moisture may accumulate to exceed the water holding capacity of waste, and to produce leachate at the landfill base.

6.2.3 PHASE III - ANAEROBIC: ACID FORMATION

Air does not readily penetrate landfills, so when oxygen is depleted, which may take only a few hours (Myers *et al.* 1979 in Lu *et al.* 1985), it is not replaced; rather anaerobic conditions are established. Strict and facultative anaerobic bacteria hydrolyse the larger organic molecules into simple carboxylic and amino acids and acid salts, predominantly acetate (Lema *et al.* 1988). *Clostridium*, *Bacillus*, *Lactobacillus*, and *Propionibacterium* were the major anaerobic metabolising genera isolated from fresh refuse, whilst the latter three genera were found in older refuse in a study reported by Barlaz *et al.*(1990). Enteric organisms, *E. coli* and *S. faecalis*, were found to significantly decline in numbers from the fresh to older waste in the same study.

The effect of microbial metabolism is to break proteins down to branched chain fatty acids, whilst lipids (fatty acid esters) hydrolitically degrade to volatile fatty acids, maximising the total volatile acid (T.V.A.) production levels. Chemical oxygen demand (C.O.D.), which reflects the amount of organic matter, is maximal following the reaction:

$$\underset{\text{organics in solid waste}}{5\,(CH_2O)X} \xrightarrow[\text{no } O_2]{\text{anaerobes}} (CH_2O)X + 2\,CH_3COOH + \underset{\text{organic acid}}{} \text{ENERGY} + 4X$$

Lignins, which are resistant to anaerobic degradation, increase proportionately in the remaining waste but act as an important adsorption and complexation medium. The polar nature of the acids produced mean they are water soluble, and enter the leachate, increasing the ionic loading of the fluid and become carried with it beneath the landfill. Where the leachate is not toxic to fungi and other bacteria, they aerobically metabolise the acids to CO_2 and H_2O. The acids give the leachate its distinctive sharp odour, increase the rate of chemical activity in the landfill, and decrease the sorptive capacity of the refuse. Large quantities of inorganics (Cl^-, SO_4^{2-}, Ca^{2+}, Mg^{2+}, Na^+) are obtained from the leaching of readily solubilised materials.

Carbon dioxide dissolves in groundwater, acidifying it, and raising the solubility and possible complexation of metal species, resulting in the concentration of metals far in excess of their normal solubilities (Lu *et al.* 1985). Complexation occurs particularly with sulphide, available from sulphate reduction, which competes successfully against other available ligands. Hydrogen gas is also released in minor amounts. As a consequence of these processes, electrical conductivity is also raised during this phase, which may last several years (Hoeks and Borst 1982 in Devlin 1990).

6.2.4 PHASE IV - ANAEROBIC: METHANE FERMENTATION

Anaerobic methanobacteria (which degrade organic matter by methane fermentation), establish only under restricted conditions, as outlined by Lu *et al.* (1985) and involve:

1) a temperature range of 10 to 16°C;
2) the absence of oxygen;
3) suitable C:N ratios;
4) a pH range 6.6 to 7.4; and
5) sufficient available nutrients.

Therefore the stabilisation phase will proceed only on the achievement of these prerequisites. The pH must increase from the acid conditions of the previous phase to near neutral levels characteristic of bicarbonate buffering systems, and substances toxic or inhibiting to methanobacteria such as Ag, Cu, or Hg must be absent. *Methanobacterium* can tolerate arsenates but are affected by some hydrocarbons such as benzene, 24-D and 24-T, and some pesticides and surfactants (Matthess 1982). These substances act as fermentation poisons and block the enzymatic breakdown and respiration chain.

If methanobacteria become established, usually within 1 year following refuse placement, they utilise organic acids to form methane:

$$5\ CH_3COOH \xrightarrow[\text{no O2}]{\text{anaerobes}} 2\ (CH_2O)_X + 4\ CH_4 + 4\ CO2 + \text{ENERGY}$$

Theoretically methane production of 0.5 m^3/kg waste is possible (Robinson 1986). Baedecker and Back (1979) suggest direct hydrogenation of CO_2 as a likely methane production mechanism; with surplus CO_2 degassed or retained as bicarbonate, but with low $H_2(g)$ levels this is a minor methane souce.

Nitrogen dominates gas output in previous stability phases but because nitrifying bacteria require some oxygen and are intolerant of high organic carbon concentrations, nitrification does not occur at this stage. Rather, denitrification occurs with the activity of heterotrophic bacteria. Amino acids are therefore broken down to other organic acids, ammonia and CO_2. Ammoniacal-nitrogen represents the ammonium ion which migrates with leachate, raising the fluid pH and either attenuates by sorption on solid materials such as clays by ion exchange, or is oxygenated in water below the landfill to form nitrates. In the methane fermentation phase, the complexation and precipitation of metals proceeds, but with increased gas production, leachate strength dramatically decreases.

6.2.5 PHASE V - FINAL MATURATION

With further stabilisation and decomposition of readily available organic matter, gas production ceases as microbes become nutrient starved. More resistant organics may be attacked by microbes which produce humic-like substances capable of complexing with, and re-mobilising, heavy metals. Oxygen and oxidisable species slowly appear and redox potential increases.

This phase may take several years to decades to complete with slower decomposition noticeable in extremely dry or saturated landfills (Lu *et al.* 1985).

Whilst this discussion incorporates gas production in landfills, the project emphasis at the Paparua County Council Landfill is on short term characterisation of the chemical nature of the leachate with regard to groundwater. Hence aqueous chemical indicators are targeted, with the expectation that they would reflect the dominant stabilisation phase at the site.

6.3 METHODS AND MATERIALS

There are two traditional approaches to the study of naturally developing leachate (as opposed to artificially developed):

1. field collection of in-situ leachate samples; or
2. production of leachate extractant samples from collected solid waste.

Field collection of in-situ leachate samples is part of the assessment of local leachate generation, constituents and migration, whilst extractant testing is used to model the sorption / desorption processes affecting contaminant release and transport under

controlled conditions. No examples were found in the literature where both approaches were used and compared to evaluate migration processes at a landfill site, as undertaken in this investigation.

Field collection of in-situ leachate was required to supplement and clarify contaminant information gained at the P.C.C. Landfill using geophysical techniques. Eight sites were selected for the installation of monitoring wells which were emplaced by the Canterbury Regional Council drilling rig. Well placement was to adequately encompass and characterise the profiled area, both at geoelectrical contaminant "hot spots" and outside waste deposition boundaries. Well construction and installation details are given in Section 4.3, with positions of the final 16 monitoring wells shown in Figures 2.7 and 4.9.

In the leachate extractant approach, collected solid waste is mixed with an equivalent weight of distilled water or other solvent, and left for a period of 16 to 24 hours (Ham 1976), or longer, to allow contaminants to desorb from the refuse. The analyzed fluid components are therefore related to the amount of waste (in units of mg/kg dry weight refuse) to estimate the quantity of specific chemicals leached from a landfill. In Noonan *et al.* (1977) waste samples from Christchurch landfills were separated into paper and general refuse components to assess differences in the behaviour of each material and test the hypothesis that paper acts as a delaying mechanism in the leaching of soluble species from refuse, preferentially adsorbing ions released from other waste. In that study waste was soaked in water-filled plastic buckets for a period of four days, with temperatures kept at 10°C. For comparative purposes, this procedure is followed for parts a and b of the first of three hydrogeochemical sampling experiments undertaken at the P.C.C. Landfill which looked at the differences in ion retention and desorption of waste.

6.3.1 EXPERIMENT 1a

During the installation of well M36/4017 at the P.C.C. Landfill, waste from the 2.7 to 5.2 m depth sampling interval was collected and separated into paper and general refuse categories. Much of the sample had not been in contact with fluid for some time. Samples were split into two: one subsample was leached in double distilled water, and the other in 0.1M acetic acid buffered to a pH of 7 with NaOH. The latter was to more readily represent the conditions found at the landfill base, where organic

acids are released from organic matter, which in turn assists in waste breakdown. Extractant fluid was decanted and bottled for analysis. Chemical analyses are presented and discussed in Section 6.4.1.1.

6.3.2 EXPERIMENT 1b

Trace element analysis by X-ray fluorescence (XRF) is a standard technique for assessing elemental content of rocks but can be applied to other substances including vegetation and food sources. In this case it was applied to newspaper samples collected from the P.C.C. Landfill to assess elemental retention changes from above to below the water table.

Five newspaper samples were analysed. Paper from the 0 to 2m and 2 to 5 m intervals in well M36/4017 were produced in 1976 when oil based ink were used by The Press company. Present newspapers use similar newsprint but are imprinted with water based ink ("The Press" Printing Room, *pers. comm.* 1989). Therefore a "Press" copy (Thursday 29 June 1978) stored since publication, was used as a control sample. Duplicates of a Tuesday 31 May 1988 issue were used to check the reproducibility of the technique since insufficient mass of the other samples was available. All samples were soaked in distilled water as for experiment 1a, then oven dried and shredded in a portable grinding mill to homogenise the sample. The paper fluff produced was mixed with a 7% aqueous solution of polyvinyl alcohol and pressed into 30 mm diameter powder pellet molds. Pellets were analysed in a Philips PW 1400 X-ray Spectrometer which used standard calibrations in the XRF laboratory of the Department of Geology, University of Canterbury (S.Brown *pers. comm.* 1990). Results are discussed in Section 6.4.1.2.

6.3.3 EXPERIMENT 2

With the installation of a second logged well (M36/4302) where sampling by rotary percussion is more controlled, samples from both the refuse and underlying sediments were collected to observe chemical changes through a vertical leachate extraction profile. In this manner the downward migration of leachate constituents is assessed. Samples were soaked in distilled water over a period of 24 hours.

6.3.4 EXPERIMENT 3

All lithologic logged wells (M36/4017, M36/4018, M36/4120, M36/4302) and

monitoring wells installed and developed at the P.C.C. Landfill, plus one well existent prior to this investigation (M36/3862), were pumped to obtain water samples. Six wells (M36/4313, M36/4314, M36/4315, M36/4311, M36/4319, and M36/4320) attested to the existence of an underlying confining silty sand by ending in that material. However, the sediment was too impermeable to allow well development, or water sampling, and these wells were subsequently useable only as piezometers. Pumping was unsustainable even at the lowest discharge rate in wells M36/4309 and M36/4310 because they ended very close to the water table. Well M36/6285 supplies water direct to a header tank for the Greyhound Clubrooms, and was unusable for geochemical sampling.

For 11 of the remaining wells, water samples were obtained through a downhole diaphragm foot valve connected by polythene pipe to a surface centrifugal pump. Wells were purged for 15 minutes before sample collection. Conductivity, total hardness, pH, and 22 inorganic cation and anion analyses by DSIR Water Chemistry Division, Ilam (July 1991) are given in Appendix 6.1, whilst TOC content analyses by the author are outlined in Appendix 6.2. These results were used to select suitable wells to test for organic contaminants in groundwater. Seven organic contaminant samples were collected in September 1991 and were analyzed for aliphatic, polyaromatic, and halogenated hydrocarbons, as well as organonitrogen, organochlorine and organophosphorus pesticides, and the BETX group: benzene, ethylbenzene, toluene, and (ortho-, para- and meta-) xylene.

6.4 RESULTS OF INORGANICS SAMPLING

6.4.1 REFUSE SAMPLING

6.4.1.1 Experiment 1a

Table 6.1 lists extractant testing results from Experiment 1 along with a summary of values obtained from extraction testing of Christchurch landfills undertaken by Noonan *et al.* (1977). The latter results attest to the inhomogeneous nature of sampled refuse; nonetheless, the range of water extracted constituents encompasses the values found at the P.C.C. Landfill. Whilst the small sample size from a single well restricts the extrapolation of extractant results, and does not allow regression correlations between variables, several features of the P.C.C Landfill samples are notable.

Desorption ratios, which relate the various combinations of solvent type and extractant

material, are listed in Table 6.2. Paper desorbed considerably more ions per unit weight than refuse, in both distilled water and acetic acid, with hardness and conductivity levels elevated in the resulting extractant. Iron and Mn are exceptions to this pattern with paper desorbing less than refuse in distilled water. Noonan *et al.* (1977) observed similar results, but found Ca and Cu were also retained longer on the paper medium. In samples obtained from the P.C.C. Landfill, paper readily desorbed Ca, Mg, Fe and Mn in acetic acid.

In the organic solvent, the release of Ca, Mg, and NO_3-N from paper were enhanced, but SO_4, Cl, and Br anion desorption were inhibited. Acetic acid promoted the release of NO_3-N and Fe in refuse samples, whilst Mg, Cl, Br and SO_4 levels were reduced. Other elements in refuse desorbed at similar levels regardless of the solvent type.

The presence of organics in the solvent most strongly affects the release of four elements: Ca, Mg, Fe and Mn. This effect is more noticeable with the paper samples and probably results from cation exchange, with Na in solution with acid salts replacing and freeing other cations from the holding positions on the paper surface. With a reduced surface area, mixed refuse is affected to a lesser degree by this process. In distilled water, Ca is the only cation more readily found than Na from paper extraction (Table 6.3), whilst in refuse Ca and Fe are preferentially liberated relative to Na. Trace metals if present, were strongly held on either paper or mixed refuse, and were found in the extractant at minor and sub-detectable levels.

To summarise; paper may initially hold more ions than general refuse, or may release them more readily where contacting solvents allow for the interchange of cations. Iron and Mn display elevated levels in refuse probably because they constitute more of that material. Although paper constitutes a considerable (and growing) proportion of refuse, the stability of cations which become held on the paper surface is transient. Both water and weak organic acid solvents, such as that found at the base of landfills, readily facilitate the removal of cations, bringing them into solution and therefore instigating contaminant transport.

6.4.1.2 Experiment 1b

Results from Experiment 1b are given in Table 6.4. The reproducibility of the technique is within XRF precision errors except for the elements Cl and Al, with

sample variation <150 mg/l. The 1978 and 1988 samples had similar elemental compositions except for Na and Cl, which are 2 to 3 times higher, and sulphur, which is 2 to 3 times lower, in the 1988 sample. Variation in these values may result from the change in ink type.

The most significant features found in paper from above and below the water table are outlined below. Both newspapers removed from landfills show considerably elevated Si and Al levels. If these are taken to represent clays retained on the paper surface, then the samples can simply be described as dirty, with the shallower, 0 to 2 m sample, three times "dirtier" than the 2 to 5 m sample. However, Ca, Na, and S content is increased in the deeper, less dirty, sample by 87%, 114% and 64% respectively. Other elements vary by only several hundred mg/l. Whilst Ca and Na were readily desorbed into extractant water (experiment 1a) sufficient quantities of these elements are still retained on paper in the leachate zone. The ready desorption of K, found at 10 times higher concentrations in paper extractant than refuse extractant, is here confirmed by the halving of K content in paper below, as opposed to above, the watertable. Whether these trends are a function of cation exchange on the clay sites, or on paper itself could not be determined by this technique. XRF may however be a suitable technique to further pursue elemental movement in refuse types and sediment.

6.4.2 TRILINEAR PLOTS

6.4.2.1 Extractant Samples

Trilinear (Piper) diagrams have been used extensively to illustrate mixing or migration trends in a sequence of water samples. Trilinear diagrams for water and extractant samples, obtained in Experiments 2 and 3, illustrate major cation and anion percentages (Figures 6.2 and 6.3) and derive from full hydrogeochemical results which are given in Appendix 6.1. All, save three, extractant values from M36/4302, plot on, or close to, a straight line, attesting to a simple mixing relationship between end members (Figure 6.2) (Matthess 1982, Hem 1985). Samples derived from within refuse (<6 m depth) exhibit higher Na + K and SO_4 content than other samples, and plot on the extreme right of the defined line. These samples fall into a Na-HCO_3-Cl-SO_4 water classification (Davis & De Weist 1967).

On the extreme left of the mixing line, samples extracted from below refuse material in silty sand (depth >11 m) plot with higher Ca and bicarbonate. The sediment from

which the extractant was derived has been shown to be relatively impermeable in comparison to alluvium both above and below this level, suggesting that landfill leachate would not easily percolate through this material. On this basis, these extractant samples relate exclusively to the sediment and do not reflect leachate.

Analyses of extractants derived from gravels lying between refuse and silty sand (Sample 7), fall midway between the end members. Hence a mixing continuum exists between extractants released from refuse and those originating from low permeability, uncontaminated, fine sediment.

Values from well M36/4017 (Experiment 1) for extractants from paper and general refuse (data points 20 and 21 respectively), also plot on this line; however they are more Ca + Mg and HCO_3 rich than refuse samples from M36/4312 and plot further to the left. This may be a function of refuse variability. Alternatively the high gravel and sand content in Experiment 1 samples may mean that elements preferentially attached to refuse are diminished with the reduced number of bonding sites found in the alluvium.

Three samples, 13, 14, and 6, are anomalous, and fall on the boundary known for > 90% of potable water analyses (Figure 6.2). These samples all have cation / anion balances in excess of the acceptable limit of 5%, and their plotting position is therefore unduly affected by specific anion species (Hem 1970). Values of pH are lower than the 6.0 to 7.8 range found for other extractant samples.

Extractant samples 13 (depth 12.25 to 13 m) and 14 (13 to 14 m), are high in sulphate and exhibit low pH values (4.3 and 5.5 respectively). These features are attributable to the peat material found within the blue silty sand at this depth. Sulphides from anaerobic peat lenses are exposed and oxidised in the soaking process, releasing additional H^+ ions into the extractant solution, thereby acidifying it. The inclusion of the sulphur rich, slightly acidic peat, is outside the system operating on the other samples and it is therefore divorced from the linear relationship.

Sample 6 (6 to 6.5 m depth) is extracted from gravel material immediately under refuse, and is the most conductive of those tested. The cation to anion ratio is unbalanced and the sample falls outside the mixing continuum. Dominated by

chloride, the sample exhibits a pH of 2.6. Whilst low pH values down to 5 are expected on the basis of other leachate studies (Section 6.4.4), a pH of 2.6 is anomalously low, but resampling was not possible.

6.4.2.2 Water Samples

Water samples from wells at the P.C.C. Landfill, both within and below refuse, have a more restricted distribution on the trilinear diagram (Figure 6.3) than extractant samples (Figure 6.2), and are classed as Ca-Na-HCO_3 type waters. Samples centre about the coordinate diamond 66 on the Furtak and Langguth (1967) classification scheme. Outliers to this diamond have <25 total cation or anion milliequivalents. Small acceptable ion imbalances for these samples therefore have a marked effect on the trilinear graphical plotting position. Water samples all lie between refuse and silty sand end members, on the horizontal extractant mixing continuum. The vertical extension of the water sample plots may result from changing redox conditions.

Baedecker & Back (1979) have characterised water by a nitrogen index, which is the ratio of organic and ammonia-nitrogen to nitrate and nitrite-nitrogen. Low numbers represent absence of ammonia, while high values arise from concentrated ammonia and nitrogen-containing organic compound indicative of reducing conditions. The transition from low to high nitrogen index is reflected as a downward vertical trend on the trilinear graph. Samples from M36/4329 (b), M36/4327 (d) and M36/4317 (l) all have NH_4-N concentrations greater than 80 mg/l and NH_4-N : NO_3-N ratios in excess of 10 (Figure 6.4) and plot lowest on the trilinear plot. Other samples, with <10 mg/l of any nitrogen species, are vertically higher, toward a Ca-Mg type water with SO_4 as the major cation. The pattern is not as distinctive as the Baedecker & Back example from the Army Creek landfill, because of lower overall chemical constituent levels.

The composition of water samples, although controlled by a mixing process outlined by extractant samples, is similar with regard to the ratios of various cation and anion species. The definition of polluted versus clean water, or of sampling position within refuse, below the refuse layer, or downgradient of the landfill, from a trilinear plot alone is difficult because of low constituent levels. Instead, the concentrations of various elements, not addressed by the trilinear plotting method, must be examined and are discussed in the following section.

6.4.3 SINGLE VARIABLES

6.4.3.1 Extractants

Sediment and refuse samples were collected as outlined in Appendix 4.1. In the drilling of well M36/4302, the well casing dropped suddenly into a zone of minimal resistance between 4 and 5 m, and consequently no sample was recovered for this depth interval. The sample for the 7 to 8 m interval was lost during transportation from the site. Chemical and physical parameters assessed from extractant samples for the other intervals excluding the 4 to 5 m and 7 to 8 m zones are given in Figure 6.5.

The major cations, Na and K display similar trends, with samples derived from within refuse (<6 m) exhibiting significantly higher values than sediment derived samples, except for the interval between 11 and 12.25 m. Anion values are likewise elevated at near surface levels but SO_4 declines at deeper than 6 m while Cl values decrease below 6.5 m, possibly the result of Cl ion migration into gravels immediately below refuse. Hardness (as $CaCO_3$) is proportional to bicarbonate, with elevated levels within refuse and within the 11 to 11.75 m, and 11.75 to 12.25 m intervals. Constituent hardness ions Ca and Mg are the major contributors to fluid electrolyte content as reflected in high conductivity, peaking at 650 mS/m in the 5 to 6 m interval, and declining to <100 mS/m in all intervals below 6.5 m.

Although Mn and Fe display high values (4 and 220 mg/kg respectively) for the 5 to 6 m collection zone, stronger concentrations were extracted from sediment at 11 to 12.25 m. These two elements, particularly Fe, contribute to the increase in TDS and water hardness at this depth.

Boron was found at 14 to 43.2 mg/kg in the refuse zone, but beneath the 6m level is reduced to subdetectable levels. Most minor elements (Br) and metals (Li, Pb, Cd, Cr, Cu) have high concentrations in the refuse but are particularly concentrated in the 5 to 6 m zone.

The hydrochemistry of extractant samples reflects the hydrogeologic and microbial processes influencing leachate production and migration. Near surface samples are subject to percolation from individul rainfall events which in contrast to the monthly averaged Water Balance results for the Canterbury climate may be sufficient to produce leachate. Groundwater levels, prior to 1987, had risen to within several

metres of the well cap level, but in later drought years lowered considerably, and were at the 5.4 m level when samples were collected. The extraction process was therefore the first occasion that near surface samples had contacted water for some time. Consequently near surface samples readily desorbed and released ions into the extractant solution. The lower part of the 5 to 6 m interval is the zone of active leachate production with refuse in prolonged contact with groundwater and therefore exhibits high ion concentrations including trace metals. Metal mobility is enhanced by low pH conditions caused by the generation of CO_2 or organic acids. The low pH value of 2.3 found in the 5 to 6 m interval appears abnormally low and may not have resulted from this processes but from the change in redox charge of ions in the sampling process. However, resampling was not possible.

Active refuse degradation is indicated by the nitrogen species distribution. Nitrate-nitrogen is found in extractant samples from M36/4302 down to 6 m, but at an order of magnitude less than the ammonium-nitrogen concentrations. Ammonium is produced by the breakdown of organic acids with a consequent increase in pH. This is consistent with extractant results with ammonium-nitrogen found at elevated levels in the refuse zone down to 6 m. The NH_4-N value of 420 mg/l in the 5 to 6 m interval distinguishes this as an active anaerobic organic matter degradation zone where nitrifying bacteria are inhibited by the high organic carbon content (cf. Baedecker & Back 1979). The downward migration of the ammonium ions in the 6 to 6.5 m and 6.5 to 7 m intervals is apparently inhibited. Low NH_4-N levels but high NO_3-N values in this 6.5 to 7 m zone indicate the oxygenation of ammonia in groundwater immediately below the landfill base in sandy gravel, but in deeper finer sediment neither form of nitrogen is found at significant levels.

The underlying low permeability sandy silts and silty sands, with the exception of the 11 to 12.25 m interval, exhibit minor levels of all chemical parameters and have therefore been unaffected by, or have not retained leachate.

Results for the 11 to 12.25 m interval are anomalous with respect to the main hydrochemical trends. Iron, Mn, Ca and Mg, which are elements often involved in cation exchange with clays, have high concentrations in this interval. In addition, the highest clay content (up to 15%) is exhibited in sediment from this sample depth. Clays were responsible for high values of these cations in the Army Creek Landfill

investigation by Baedecker & Back (1979) and are further discussed in Section 6.4.5.

6.4.3.2 Water Samples

Results of inorganic analyses on water samples are presented in Figure 6.6 and reveal mixed patterns of water quality. pH is fairly uniform for all samples, between 6 and 6.5, but conductivity is notably higher in samples from M36/4327, M36/4329, M36/4317 and M36/4316 (300, 240, 230, 160 mS/m respectively). For discussion purposes these four wells will be referred to as "Group 1" wells. Samples from Group 1 wells exceed Guideline values for Cl and Na, with values of >100 mg/l, and exhibit significantly higher hardness values than found in other wells, even though all samples exceed the 200 mg/l Guideline value. Similarly whilst no guidelines exist for Ca, Mg, conductivity, TOC, Sr or Br, Group 1 wells show considerably elevated levels of these parameters relative to other wells. High TOC values indicate the presence of volatile fatty acids, carbohydrates, amino acids and alcohols. Group 1 wells also exhibit a total dissolved solids range of 1238 to 1736 mg/l and rate as brackish water (cf. Davis & DeWiest 1967), whilst all other samples have TDS values between 200 and 655 mg/l.

Most wells exceed the Guideline of 0.5 mg/l for Mn; but Mn is highest in wells M36/4316 and M36/4317 at 7.8 mg/l and 9.0 mg/l respectively, and <2 mg/l for all other samples. Similarly Fe is found at elevated levels in all wells but is significantly higher than in other wells, at 95 mg/l in M36/4316 versus <40 mg/l Fe for all other wells.

Sulphate levels of 190 mg/l from M36/4316, and 68 mg/l from M36/4312 both exceed the Guideline value of 50 mg/l. At the Borden Landfill in Ontario, high sulphates have been attributed to gypsum in wallboard and plaster, that have been disposed of in the landfill (Nicholson *et al.* 1983, Cherry 1981). At two similar sites, Woolwich and North Bay Landfills in Ontario, low sulphate levels were found as a function of the reducing conditions and this is taken to be more usual (Cherry 1981). High sulphate values, particularly in Well M36/4316, therefore appear anomalous.

Nitrate nitrogen levels do not exceed the Guideline value of 10 mg/l in any well samples but three of the Group 1 wells (M36/4327, M36/4329, M36/4317) display significantly elevated ammoniacal-nitrogen levels, up to 160 mg/l, whilst all other wells have values <10 mg/l. The ammonia in these three wells is evidence of microbial

degradation of organic matter. A similar hydrochemical pattern is illustrated by potassium and boron content, although for the latter element a fourth well, M36/4120 (slotted throughout the 7 to 30 m depth), also has high levels.

Wells M36/4312 and M36/4316 are high in Li (0.04 and 0.07 mg/l respectively), whilst M36/4327 and M36/4329 with values of 0.02 mg/l, also exceeds the Li Guideline level of 0.01 mg/l.

The Guideline health level for Pb (0.05 mg/l) was exceeded in only one well: M36/4316 at 0.12 mg/l. This well also had a high Cu level (0.11 mg/l), as did well M36/4325 (0.08 mg/l). Zinc levels were elevated above that found in other samples in two of the Group 1 wells, M36/4327 and M36/4329, but were below Guideline levels. Low heavy metal levels are consistent with other studies. Young *et al.* (1984) found in laboratory tests loading a 0.5 m column of pulverised municipal waste with a solution of metals (Cd, Cr, Cu, Fe, Pb, Hg, Ni, and Zn) that only Ni was not wholely absorbed from solution. Borden & Yanoschak (1990) found that the majority of heavy metals were below the analytical detection levels in their study of leachate from 71 landfills, whilst Clark & Piskin (1977) found heavy metals (Zn, Cu) and boron to contribute $<1\%$ of the total leachate composition in 123 leachate samples from 54 landfills in Illinois. Of these metals, only Zn was found at detectable levels in groundwater beneath each of the Borden, Woolwich and North Bay Landfills sited upon sandy glaciofluvial aquifers in the Cherry (1981) study. It is noted in many of these papers that Zn may derive from landfill emplaced cosmetics.

Five wells, M36/4017, M36/4302, M36/4325, M36/4308, and M36/3862, will be referred to as "Group 2" wells. Group 2 wells are "clean" in terms of inorganic constituents except for Fe and Mn, and by extension hardness, for which the Highest Desirable Guideline of 80 mg/l was exceeded in all the wells surveyed. Levels of Fe and Mn which exceeded Guideline levels have also been found at the Borden Landfill and were recognised by Nicholson (1983) to be impervious to adsorption effects. High Fe and Mn levels have been found in groundwater from the general Wigram vicinity and these wells are therefore considered to be unaffected by leachate and serve as "background" water analyses. None of these wells had TOC values in excess of 35 mg/l. As such, these wells illustrate that the extent of groundwater contamination is constrained both laterally and with depth.

Well M36/4308 provides evidence of the constraint of contamination for the northwest of the landfill; with no contaminants found in groundwater at 6.5 m beneath the surface. Likewise, groundwater sampled from M36/3862, at a depth > 8.5 m shows that leachate is absent in the southwest. Wells at BORE, the southeastern-most sampled wells, reveal interesting hydrochemical patterns. M36/4325 was driven to 8.5 m and shows elevated Cu and Pb levels relative to M36/4324, positioned one metre shallower. Of these elements, only Cu exceeds the Guidelines. M36/4017 and M36/4120 in the same well cluster are slotted from 7.5 to 30 m, but water samples from these wells are free of Cu, Pb and Zn. In the northeastern well cluster, M36/4317 at 6.5 m and M36/4320 at 8.5 m are both contaminated with a variety of elements but M36/4302, at a depth of 16 m, is untainted in all but hardness.

These results show that the contaminating substances have not moved downward and are restricted to water shallower than 9 m and within the general refuse deposition boundary. This hydrochemical pattern is coincident with the 56 Ωm zone determined by resistivity profiling, and confirms the tabular nature of contamination seen on pseudosections; fine sediments restrict downward percolation. Whether contaminants are typical of leachate contamination and are derived exclusively from refuse contacting water is examined in the following sections.

6.4.4 SINGLE PARAMETER INDICATORS

Due to high analysis costs, researchers have often recommended single parameters which are consistently higher than water quality guidelines, to be used as stand-alone pollution indicators, where at a given place and time their higher values are not attributable to geologic origin (Matthess 1982). Indicators must also be readily available in a variety of waste types, mobile and persistent in groundwater, at contrasting concentrations with background groundwater, and be able to be simply analysed (Clark & Piskin 1977). MacLean (1969) and Davison (1969) suggested K but this was denounced by Matthess (1982) because potassium is strongly sorbed below the surface and does not migrate with the polluted groundwater; instead HCO_3, Cl, NH_4, NO_2 were recommended. Other useful pollution indicators have come from Kimmel & Braids (1974): Cl, Na, Fe, dissolved metals and CO_3; Baedecker & Back (1979): Fe and Mn; Clark & Piskin (1977): B, Fe, NH_4, and Cl and hardness (in some circumstances); and Borden & Yanoschak (1990): Zn, TOC, conductivity, TDS, Pb and heavy metals. Dance & Reardon (1983) indicate that in the migration of a

contaminant with equimolar concentrations of the major cations, Ca is liberated by ion exchange between the aquifer materials and the tracer producing a "hardness halo" at the leading plume edge. Calcium was the dominant cation in the Borden Landfill plume (Nicholson *et al.* 1983). Although this was in a carbonate aquifer a similar effect may occur in other environments. Whilst findings are often site specific, it is useful to compare the typical ranges of leachate constituents in order to gauge the most useable single pollution indicators and thus set priorities for constituent analyses.

A compilation of leachate compositions found in a variety of both New Zealand and international studies is shown in Figure 6.7. The presentation format is adopted from Lu *et al.* (1985) who provided data for 13 studies (30 samples). The data are listed according to the key in the accompanying Table 6.5. Seventeen further sources (26 samples) including New Zealand studies, and the authors sampling at the Paparua County Council Landfill have been added. In several cases the sources were themselves compilations of results from various landfills, rather than a range for a single site. Examples of synthesised leachate behaviour from test cells, lysimeters, laboratory cells/columns, and/or artificial leachate mixtures have been omitted from Figure 6.7 but would have fallen within the range found from these determined values (see Lu *et al.* 1985). All data shown is from landfill sampling, either within refuse, the underlying groundwater, or leachate collection systems. Several studies list samples from nearby upgradient or downgradient streams, ponds or wells, for background contrast. Only data where 'leachate' is described or the connection between the sample and the landfill clearly made, are included although sampling circumstances are often poorly reported. Thus, the lower, more dilute, background values are truncated from the diagram. In several instances, samples contaminated in various ways, for example by seawater in Moss (1988), are omitted. Samples 30 to 34, although shown, are suspected to have been collected through, and contaminated by, soil (Lu *et al.* 1985).

Landfill age, and setting, for each study is outlined in Table 6.5; however studies commonly report "old" or "new" landfill or refuse ages without explanation as to what these terms mean. Likewise, not all studies report either the same components, or use a standard presentation of each. For example, nitrogen can be reported in either separate, or a combination of nitrogen compounds. Substances such as Hg, F, or phenols, are omitted due to data paucity. Where studies report both "average

composition range" and "total range" for a number of analyses, the latter is shown. In addition, the total range is summarised from studies where sampling is from a number of locations over a time span. This obscures much of the seasonal and local variation (described e.g., in Rogers & Totton 1988), but as refuse and leachate exhibit broad inherent, and reactive, variability within such diverse settings, the detail may be lost in comparing only two sites, let alone the number presented. Figure 6.7 therefore shows the total range of landfill-sampled leachate for broad comparative purposes only. Potable water aesthetic and health guideline values from New Zealand, the World Health Organisation, and the National Academy of Science and Engineering (USA), are superimposed. In addition five parameters (pH, COD, Cl, Fe, Mn) used by Rogers & Totton (1988) after Prati (1971), as single pollution indicators have the associated pollution levels displayed. New Zealand studies, and P.C.C Landfill results are separately identified for comparative purposes.

Several trends are apparent. Most obviously leachate compositions are highly variable as a function of site specific factors with some chemical constituents and water quality parameters (conductivity, Cu, Zn, Mg, hardness, SO_4) ranging in excess of 5 orders of magnitude. New Zealand leachate results are useful as indicators of ranges within sites, with many of the other studies showing peak arithmetic mean, or singular values. The maximum range for any single parameter, within a specific site, rarely exceeds 2 orders of magnitude.

Many of the maximal values are from compilation studies, and it is difficult to know if only 1 or several samples or sites, are this extreme. Maximal values for copper at, for example, 1100 mg/l is almost 2 orders of magnitude greater than the next highest reported value. Likewise Fe, Mn, conductivity, Cd, SO_4, and Ni all have maximal values at least 1 order of magnitude in excess of other results. Additionally, New Zealand leachate analyses lack the higher levels of constituents as found in other studies, but otherwise show similar results. Both features are probably due to concentrated industrial waste disposal found in some overseas landfills and lacking in New Zealand. Leachate derived from the Paparua County Council Landfill consistently falls amongst the lower reported values for all parameters.

For several parameters the majority of studies shown reported values in excess of guidelines shown. These include major cations and anions: hardness (100% > NZ

Excessive Standard), Na (78% > NZ Highest desirable range), Ca (80% > NZ Maximum acceptable), Mg (87% > NZ Maximum acceptable), Mn (89% "slightly to heavily polluted"), and Fe (96% "heavily polluted"). For the metals, 81% of reported values for Ni are in excess of detection levels, all save one value exceeds health standards for Pb, whilst Cu, Cd, and Cr values range markedly but the majority exceed the lower of the health standards shown. For Zn, 15 out of 31 reported studies include values in excess of NZ maximum acceptable standards. By contrast only 7% of studies reported any samples having pH falling outside the 5 to 9 range; the acid and alkali extremes are drawn from compilation studies. All New Zealand leachates, including the P.C.C. Landfill samples, fall within the "acceptable" pH range of 6 to 8.5.

There are problems with the application of most of these inorganic parameters. Sampling points close to seawater or road salt application sites (not usual in New Zealand), can give misleading Cl or Na values. Likewise Mg, Ca, and by association hardness, have naturally elevated levels in carbonate aquifers, whilst Fe and Mn levels are high in some sediments and are associated with microbial action on well screen and casings (Figure 6.8). Ca, Mg, Mn and Fe are all elevated in extractants from, and water samples obtained through, fine grained sediment underlying the P.C.C. Landfill.

Boron in contrast, is rarely found naturally occurring except in argillaceous marine sediments, evaporites and thermal water and associated deposits (Hem 1985, Schofield 1960, Goldschmidt 1954). It is however, common in sodium tetraborate (borax), a cleaning aid and flux, and in silicic borates from glassware. Boron is also used as a preservative, fire retardant and glaze, and *"hardly any other substance enters so many diversified lines of manufacturing"* (Jensen & Bateman 1981), all of which will be disposed of in landfills. In all studies where boron was tested for in leachates, including that of the Paparua County Council Landfill where values ranged from 2.3 to 2.9 mg/l, it is found to exceed the Guideline for plant use. Boron should therefore be more frequently incorporated into leachate analyses, as background levels in most groundwater sampling settings, including the Canterbury Plains groundwater, will be negligible.

Given the chemically inert nature of the Canterbury aquifer sediments Cl would be unaffected by adsorption mechanisms. It is the only ion found in extractant testing at high levels in the gravel immediately underlying refuse, and would therefore be at the

forefront of a migrating plume. Calcium is known to be released from cation exchange sites at the expense of other cations and creates a "hardness halo" at the leading edge of contaminant plumes even in aquifers with calcite and dolomite content (Cherry 1981). Therefore Ca and Cl, in addition to B, would be useful single parameter contaminant indicators in the Canterbury setting.

6.4.5 OTHER CONTROLS ON SAMPLE CHEMISTRY

Sediment obtained from the 11 to 11.75 m and 11.75 to 12.25 m intervals in well M36/4302 exhibited elevated cation concentrations in water extracted samples which may result purely from clay particulates in the water sample. However, a change in the sediment type is also suspected. Wood obtained from well M36/4017, was found at a level equivalent to the 13.165 to 13.665 m depth in M36/4302. The carbon dated age, approximately 8820 ± 70 B.P. years, is just prior to the maximum landward incursion by the sea known for the Canterbury area. The increasing clay and silt content at this depth and evidence of peat and wood chips indicate swampy deposition conditions. Whilst quartz and feldspar sands and gravels appear ubiquitously throughout the profile, the clay type was undefined. The elements found at high levels in anomalous positions, Ca, Mg, Mn, and Fe, are typically involved in cation exchange on clay surfaces and interlayers. If marine swelling clays exist in these intervals, which due to low permeability had not been "flushed" by groundwater, it is possible that contact with distilled water in the extraction process has allowed cation exchange to occur. Cations from the clay surface and interlayer positions would be flushed into solution, replaced by H^+ ions. To pursue the possibility, both untreated and glycolated clays, from the 11 to 11.75 m, 11.75 to 12.25 m, and the bracketing sample intervals, have been analysed by X-ray diffraction. Glycolation changes the position of characteristic XRD peaks allowing the presence of swelling clays to be determined.

XRD graphs for samples 10 to 11 m and 11 to 11.75 m were almost identical, as were those of samples 11.75 to 12.25 m and 12.25 to 13 m. Only two graphs are therefore given in Figure 6.9. Below 11.75 m the clay fraction is dominated by feldspars, albite and anorthite (or variants with similar XRD patterns), and quartz. Shallower than 11.75 m, mixed clays, kaolinite, chlorite and illite are present along with feldspars and quartz. Glycolated XRD traces replicated untreated sample traces therefore no montmorillonite or vermiculite swelling clays are present. Because the fine sediment mineralogy is distinct between the samples, it does not correlate with the extractant

hydrochemistry trends which are similar for both samples but different from adjacent samples.

Variation in clay type does not therefore definitively account for the high cation extraction in values at the 11 to 12.25 m depth. Regardless of the cation release mechanism, this zone is distinct from leachate in the landfill, and has not resulted from downward percolation. High chemical constituent levels can therefore result from sedimentological factors or poor sampling and must be distinguished from anthropogenic contamination to avoid the misinterpretation of water and extractant sampling.

6.4.6 BIVARIATE RELATIONSHIPS

Whilst the relationships of various parameters to water quality guidelines indicates those samples which are contaminated, it is not possible to know if the contamination is from leachate, other artificial sources, or is in fact a natural characteristic of the aquifer. Siegel *et al.* (1990) advocate a geochemical process approach, using bivariate ion scatterplots based on the prediction of probable chemical reactions in the groundwater environment. For example stoichiometric proportions of the most soluble minerals in the Vermont Shale, Na vs Cl (halite) and Ca vs SO_4 (gypsum) in groundwater are linearly related, unaffected to any great degree by other ions. In that study, leachate samples plot as for groundwater on the Na vs Cl graph, but as outliers to the groundwater Ca vs SO_4 relationship. Leachate composition was therefore subject to an additional control on Ca and SO_4 ratios, and the bivariate graph was used to distinguish samples derived from outside uncontaminated groundwater, distinguishing whether this was leachate, a secondary water source or other artificial contamination.

For the P.C.C. Landfill, known leachate compositions are given by extractant values. Because parameter ratios are used, extractant values are directly comparable with in-situ water samples. Thus, not only should outlier identification be possible, but confirmation of leachate as the source is available.

As Canterbury aquifer sands and gravels are composed predominantly of quartzose and feldspathic greywacke which is relatively inert in water, the distinctive ion relationships predictable in carbonate sediments do not apply. A number of both

chemical and physical parameters were compared to see if inherent leachate and groundwater groupings were distinguishable, as illustrated by the conductivity versus Mn plot. Cluster analysis may be useful in this regard with larger sample sets.

Extractant samples from well M36/4302 fall into two distinctive groups on a conductivity vs Mn plot (Figure 6.10a), one of which, the "leachate" group, comprises samples derived from within and immediately below refuse at a depth <8 m. The second group is composed of samples extracted from underlying less permeable silty sand, with samples extracted from 11 to 12.25 m plotting farthest from the origin. When in-situ water samples are superimposed (Figure 6.10b), the bifurcating pattern is enhanced. Samples from M36/4327, and M36/4329 which exceed Guidelines for most parameters, lie along the correlation line for leachates. When these two samples are included in the regression line of leachate extractants, a regression coefficient of 0.87 results. This is only slightly less than that established for extractants from <8 m only, of 0.91, and indicates that the two water samples may be leachate derived. All other water samples fall on a "groundwater" line with r=0.946 (Table 6.6). Extractant values for >8 m lie to the left of the "groundwater" correlation and are characterised by very low conductivities irrespective of Mn content.

Similar patterns emerge in many of the bivariate combinations tried, but three of the Group 1 water samples (M36/4327, M36/4329, M36/4317) consistently fall on leachate extractant lines rather than the two seen in this example. Extractants, derived from sediment deeper than 8 m, were consistently removed from the groundwater line but may be affected by additional controls as discussed earlier.

Table 6.6 lists various bivariate combinations where the regression correlation exceed 0.65 for any of the subsample groups. The bivariate relationship can be used to distinguish leachate contaminated groundwater samples from uncontaminated groundwater, where parameter correlations using all extractant and water samples is poor, but extractant samples <8 m display a good correlation. Plots of SO_4 vs Zn, conductivity vs Fe, SO_4 vs B, and TOC vs NH_4-N fit this criteria. Conversely, the Ca vs Cl plot shows a strong correlation at 0.972 for uncontaminated water samples i.e., the water sample group minus M36/4317, M36/4327 and M36/4329, but a poor correlation for leachate extractants from <8 m (r=0.108). These latter parameters directly affect each other in groundwater but are influenced by other forces in

leachate, and the relationship can be used to distinguish the sample origin.

Naturally, conductivity values are linearly related to total dissolved solids (TDS) and to many of the predominant major cations and anions.

Notably, uncontaminated water is characterised by low parameter values but the bivariate technique is more useful than separating the samples on the basis of water quality standards as discussed below. Sample M36/4316 exhibits raised levels of some chemical and physical parameters. On the bivariate plots however, it separates off from other Group 1 contaminated samples indicating different controlling factors. High in Ca, Mg, Mn, Fe, SO_4, COD and TOC, the sample exhibits a slightly lower pH value (6.1) than other water samples. It is also the only water sample to fall outside a well defined Na vs Cl relationship (Figure 6.11), and unlike samples from M36/4327, M36/4329, M36/4317, did not display yellow brown discoloration, frothing or a pungent odour. Unlike other water samples which were collected in sand and gravel, this sample was murky with grey sediment and was obtained with difficulty from a well which ended in silty sand. Chips of wood and peaty clay blebs were pumped from this well during surging. Other wells to the south of the landfilled area also ended in organic rich silty sands. The sediment in these wells was too impermeable to constantly recharge with water and wells pumped dry. It is possible that similar water chemistry would have resulted from these wells if they had yielded water samples. Borden & Yanoschak (1990) found that high Fe and Mn concentrations in landfill sampling was due to the collection of suspended clay with the groundwater sample, which is exemplified by the M36/4316 sample.

Therefore, whilst the M36/4316 water sample shows elevated levels of some constituents, it is not leachate derived. Instead, like the two extractant samples discussed earlier, the extractant chemistry is governed by factors not apparent in simple variant comparisons with water quality standards.

6.5 RESULTS OF ORGANICS SAMPLING

6.5.1 INTRODUCTION

Organic compounds in leachate result from either anaerobic biological decomposition of waste producing acids and alcohols, or are synthetic, originating from chemical manufacturing. Decomposing organics are usually problematic in groundwater due to

aesthetic colour and smell degradation. Synthetic organics have possible toxic, carcinogenic or mutagenic properties and can be persistent long term in groundwater (McGinley & Kmet 1984). However, testing of groundwater for organic contaminants is rarely undertaken because of the expense, and usually follows intensive inorganic indicator testing unless there is a specific concern about organic contamination. Shackelford & Keith (1976) in Freeze & Cherry (1979) note that in excess of 1200 manufactured chemicals have been found in groundwater. Of these, pesticides and industrial chemicals are targeted for detection because of the quantities produced and released, and the individual substance characteristics (Hem 1985). The greatest threat and therefore most highly prioritised substances are soluble, toxic, refractory and nonvolatile (although some volatile substances can have their volatility reduced in groundwater causing problems). These substances are immune to bacterial degradation and do not bind with soil or organic matter. In particular, chlorinated hydrocarbons, which are used as degreasers because of their solubility in nonpolar substances, but poor solubility in water, are commonly found in groundwater.

6.5.2 PARTITION COEFFICIENTS

In order to compare the behaviour of various organic substances in groundwater a reference coefficient (K_{oc}) is defined as follows. Substance mobility is proportional to water solubility, and inversely proportional to the sorptive power of ion exchange sites in the sediment. Substance mobility is quantified by K_d, the soil solute distribution coefficient:

$$K_d = \frac{d C^*}{d C}$$

where C^* is the mass of solute sorbed per unit of dry mass soil, and C is the solute concentration. For organics, sorption is partly controlled by the characteristics of the chemical itself, but additionally by the amount of soil organic carbon, a site specific factor:

$$K_d = K_{oc} \times \% \text{ Organic Carbon in the soil.}$$

Thus organic chemicals are described in terms of K_{oc}, the octanol water partition coefficient, being the relative tendency for the substance to remain dissolved in water or in n-octanol when shaken in a solution of the two. Representative values of K_{oc} for specific organics relevant to this investigation are listed in Table 6.7, and can be used to predict organic behaviour when contaminants do not exceed sorption kinetics by rapid transportation.

6.5.3 PREVIOUS WORK

In the few studies of organics leached from municipal solid waste landfills, there are similarities in the types of substances detected. Borden & Yanoschak (1990) reviewed water quality records for North Carolina and found that 14 of 37 (38%) active or closed (post 1984) municipal sanitary landfills, have had organics detected in one or more monitoring wells. Substances most frequently found are chlorinated solvents (3 out of 37 wells or 8.1% of the samples), petroleum derived hydrocarbons (3, 8.1%) and pesticides (2, 5.4%). The severity of contamination varies between localities, from <10 mg/m^3, to sites suspected to include concentrated industrial waste, with total organics content in excess of 1000 mg/m^3. Thus while some municipal solid waste (MSW) landfills had organically degraded groundwater, the majority of sites suffered no detectable contamination.

McGinley & Kmet (1984) present organic analyses from nine Wisconsin MSW landfills, confirming findings found by other researchers at a variety of sites in Oregon (3 sites), Minnesota (6), Colorado (1), Oklahoma (1), Delaware (1) and Michigan (1). Contaminants found in the Wisconsin sampling included many persistent, volatile and suspected carcinogens. Halogenated ethers and aliphatics (used in refrigerants, insecticides, solvents, propellants and cleaners) were commonly found. Within this later group, methyl chloride was found in 4% of samples (range <1 to 170 mg/m^3), but methylene chloride was found in 79% of samples (range <10 to 20000 mg/m^3), demonstrating the high variability of contaminants, both in concentration and frequency of occurrence. Other halogenated hydrocarbons observed include: 1,1 dichloroethane (57%), trichloroethylene (49%), tetrachloroethylene (39%), 1,1,1 trichloroethane (30%). Few polyaromatic hydrocarbons except naphthalene, and no nitrosamines or PCBs were detected. Phthalate esters, the plasticisers found in packaging, cosmetics and oils, were commonly found, as were the monocyclic aromatics, the BET group present more often than halogenated varieties. Xylenes were found at both sites tested for it in Wisconsin as well as in studies in Minnesota and Oregon. Pesticides and herbicides, as in other studies, are consistently absent (e.g., Spencer 1984) except for one occurrence of delta BHC, and two of 2,4D. Organic testing in groundwater beneath the North Bay Landfill indicates the presence of chlorobenzene, dichlorobenzene, acetone, chloroethane, dichloroethylene, xylene and naphthalene (McGinley & Kmet 1984). North Bay has accepted sewage sludge and local industry waste along with municipal solid waste during its lifetime.

In Canterbury, testing for organics in groundwater is undertaken annually at the active Christchurch Metropolitan (Bottlelake) Landfill which superseded Local Authority dumping sites in the early 1980's. Moderate mobility (low K_{oc}) chlorinated hydrocarbons, including tetrachloroethene, 1,1,1 trichloroethane and trichloromethane (chloroform), have been detected in groundwater both up- and down-gradient of a completed cell of the Landfill between 1986 and 1991, along with various polyaromatic hydrocarbons including chrysene, benzo(a)pyrene, benzo (k)fluoranthene, pyrene, benzo(b)fluoranthene, benzo(a)anthracene and fluoranthene (Boulay 1991, Close 1991). The migration of these organic substances is the focus of ongoing monitoring.

Prior to the Paparua County Council investigation, the only screening for organic contaminants around other Christchurch landfills had been sampling downgradient of the Johns Road site, in northwest Christchurch, where 1,1,1 trichloroethane was found. Preliminary groundwater screening undertaken throughout Christchurch City was also of relevance to the P.C.C. Landfill site.

Groundwater sampling from 34 widespread Christchurch City wells and riverwater sampling at 5 locations, by the Canterbury Regional Council in two priority chemical surveys (November 1988 and April/May 1989), indicated the presence of organics in shallow aquifer waters (Freeman 1989a, 1989b). Contaminants detected were mobile water soluble solvents and degreasing chemicals, 1,1,1 trichloroethane (19 wells, up to 2.8 mg/m^3), trichloroethene (3 wells, up to 16 mg/m^3), tetrachloroethene (1 well @ 0.25 mg/m^3), and trichloromethane (1 well @ 8.2 mg/m^3). No pesticide residues or cyclic hydrocarbons including the BETX group have been found in groundwater, whilst traces of azinphos methyl (an organophosphate insecticide), terbythylazine and simazine (persistent and mobile triazine herbicides) have been detected in river waters. Only chloroform exceeded the Guideline level in 1 well , which was shown by later testing to have been a shortlived contaminant occurrence. The detection of these organics is of concern because chlorinated solvents progressively dehalogenate to other often carcinogenic compounds (Newsom 1985). More obviously the presence of organics attests to the transit into and through the aquifer system, and where a single compound is detected other, as yet undetected, organic substances may also exist. Several of the wells incorporated in the C.R.C. survey (M36/916, M36/2050, M36/917) lie immediately downgradient of the Paparua County Council Landfill at Carrs Road, a possible contaminant source.

6.5.4 PAPARUA COUNTY COUNCIL LANDFILL

Screening for organics in 7 water samples obtained from monitoring wells at the Paparua County Council Landfill was undertaken by DSIR, Gracefield, using capillary column gas chromatography with a mass selective detector. Results are given in Appendix 6.3 along with detection limits for each substance. Only two samples, those from wells M36/4317 and M36/4327, contain organics at detectable levels, but both were described as organically very dirty (E.J.J. Bathhurst *pers. comm.* 1991). Detailed analysis reveal a variety of hydrocarbons including; nonvolatile alkyl benzenes such as methylethylbenzene, S- and N- heterocyclic compounds, and terpenes. Water from M36/4317 also contain amines and ketones whilst the M36/4327 sample, lower in overall organics concentration with less variety, contain tribytyl phosphate. No attempt has been made to quantify any of the compounds present except xylene which was found in M36/4317 at a concentration of 5 mg/m^3. No phthalate esters, halogenated compounds, phenols or pesticide residues were detected.

6.5.5 DISCUSSION

Notably only two wells at the P.C.C. Landfill, both sampled within the active leachate generating depth, and coincident with geophysical "hot spots", exhibit organic contamination. Wells below both refuse, and the lower confining silty sand, are clean of organic contamination, as are several wells to the west where shallower refuse emplacement occurred. The detection of amines, produced by reduction of nitro compounds or amides, and ketones, the "smelly" oxidised alcohols, attest to the natural degradation of organic matter. Ketones ferment to acetic acid and carbon dioxide acidifying leachate in the process. The presence of terpenes, known as the unsaturated hydrocarbons or essential oils of plants, similarly demonstrates the decay of organic matter which constitute a significant proportion of waste deposited in the P.C.C. Landfill. Synthetic organic substances include xylene (a petroleum distillate and solvent), unspecified alkylbenzenes, and unspecified S- and N- heterocyclic hydrocarbons possibly including pyridine (found in coal tar and used as a solvent).

Phthalate esters (plasticisers) almost ubiquitously present in operating and recently completed landfills (E.J.J. Bathhurst *pers. comm.* 1991) are absent at the Paparua County Council Landfill. This attests to the history and age of the site which was completed just at the onset of refuse collection in plastic bags and the widespread use of plastics packaging.

Despite widespread local horticultural use and subsequent disposal of residues in the Landfill, pesticides are not found in the leachate. This is congruent with findings by McGinley & Kmet (1984) and low appearance rates of pesticide in leachate found by Borden & Yanoschak (1990). Acidic (e.g., 2,4-D and 2,4,5-T) and nonionic pesticides are generally retained by soil organic matter, or more commonly are biochemically degraded. Subject to moisture, temperature, pH and microbial controls, degradation is relatively easy once the halogen bond is broken, which occurs more readily under anaerobic conditions (Newsom 1985). Pesticide adsorption onto clays is rarer and pH dependent, with pesticides normally hydrophobic and repelled by negatively charged clay surfaces (Saltzman & Yaron 1986). Thus many pesticides will readily sorb onto soil or other organic material such as refuse, but are less readily entrapped in sediments low in organic matter. Subsequently, where pesticide contamination is reported in groundwater, direct pesticide/water contact without transmittal through a soil medium has usually occurred (Matthess 1982). This accounts for the presence of pesticides in rivers in Canterbury and not in groundwater.

Alexander & Lustigman (1966) found the most persistent of the substituted benzene pesticides to be the refractory sulphonates as well as nitro- and chloro- substituted benzenes. Other chlorinated organics are highly refractory and can pass through a biologically active aquifer with little degradation (Cheremisinoff 1984). Degradation of halogenated hydrocarbons has not been observed in experiments reported in Newsom (1985). Chlorinated hydrocarbons are also "sinkers" (specific gravity greater than water) percolating downward throughout the saturated phase of the aquifer, transferring contaminants into solution faster than other organics and providing more contact area for detection. The converse applies for "floaters" such as benzene, examined in Hadley & Armstrong (1991).

With depth, the number and diversity of microbes capable of organic degradation decrease sharply. In addition biodegradation is threshold controlled so that low organic concentrations, of the order of mg/m^3, may be insufficient to support bacterial growth with a consequent slow breakdown rate (Cavalier *et al.* 1991). It is therefore not surprising that low levels of chlorinated hydrocarbons are predominant in literature studies, and are detected at the Bottlelake Landfill. The absence of chlorinated hydrocarbons at the Paparua County Council Landfill therefore signifies that either these organics were never emplaced, have yet to leak from the containers they were

disposed in, or they have already migrated offsite. Whilst migration downgradient is indicated with the detection of chlorinated hydrocarbons in the C.R.C. survey, site hydrogeological constraints illustrating lateral and depth leachate confinement, and the absence of any chlorinated residues, do not support this hypothesis.

6.6 SUMMARY

Leachate is actively being generated at the Paparua County Council Landfill with hydrochemical results confirming the distribution pattern indicated by geophysical profiling. The groundwater degradation pattern is similar for both inorganic and organic contaminants but the latter group are restricted to the active leachate generating zone along the northeast boundaries of the site. Organic contamination results mainly from the natural decay of vegetation and other organic matter producing ketones, amines and terpenes. Synthetic organics, specifically xylenes, are present, presumably due to solvent or petroleum discards in the Landfill. Phthalate esters usually found in abundance in leachates are conspicuously absent at the P.C.C. Landfill, attesting to the landfill history having been completed prior to the widespread use of plastic packaging. Pesticides and herbicides, also absent, are thought to have been rapidly broken down and bonded onto organic matter in the landfill. Chlorinated hydrocarbons have not been detected in hydrochemical samples, and were either not disposed of in the landfill or have yet to leak from the containers they were discarded in.

Leachate is similar in composition to that found in overseas leachate studies but lacks the high extreme values characteristic of concentrated industrial waste. Neglecting Fe and Mn, contamination by inorganic substances is evident in only Group 1 wells; Guidelines are exceeded for a variety of constituents. Guidelines for Cu, Cl, Na, and B are exceeded, with values up to 0.11, 270, 210, and 2.8 mg/l respectively, in water samples from monitoring wells. Whilst there are no Guideline values for Ca, Mg, conductivity, TOC or COD, four Group 1 wells (M36/4327, M36/4329, M36/4317 and M36/4316) exhibit levels of these parameters significantly higher than those found in other wells.

Leachate-influenced samples within Group 1 (M36/4327, M36/4329 and M36/4317) are distinguished from naturally contaminated samples (M36/4316) on the basis of bivariate plots. In M36/4316 the high ionic content, including Fe, Mn and SO_4, is due

to particulates collected in the fluid, a result of poor water sampling. Knowledge of the element sources, assisted in this case by extractant testing, is therefore emphasised to prevent the misinterpretation of leachate presence and migration.

The three leachate-influenced wells directly accessed the active leachate generation zone in the northwest corner of the P.C.C. Landfill and produced fluid samples which were frothy with yellow discoloration and a pungent odour.

Five Group 2 wells, M36/4017, M36/4302, M36/4325, M36/4308 and M36/3862, are free of inorganic contaminants except for Fe and Mn which are generally elevated above Guideline values in the Wigram vicinity and are known to be relatively unaffected by adsorption effects. These wells therefore illustrate that contaminant concentration decreases rapidly, both with depth and laterally. On the basis of groundwater constituents exceeding Guidelines contaminated groundwater is presently constrained to a zone <9 m deep and extends no farther than the 56 Ωm resistivity zone outlined by geophysical profiling.

Trilinear plots of extractant sampling indicate a mixing continuum between refuse and uncontaminated sediment; however water samples are relatively homogeneous with regard to cation and anion ratios despite their origin. Vertical segregation on the basis of redox conditions is only subtly evident due to the low overall cation and anion loading in the water samples.

Elevated Ca, Mg, Mn and Fe levels were also found in extractant samples from the 11 to 12.25 m depth interval in well M36/4302 which could not be accounted for by changes in clay type from fines dominated by quartz and feldspar to illite-, chlorite- and kaolinite- rich sediment. The liberation of these elements in extractants at this depth, and their elevated occurrence in water from well M36/4316, initially appear anomalous but bivariate plots of various constituents suggest that this too is a function of particulates collected with the fluid sample.

Boron, often omitted from water sampling, is recommended as a single parameter indicator of leachate contamination, along with Cl and Ca, which should indicate the furthest contaminant extend by the hardness halo effect. TOC indicates leachate affected groundwater much more clearly than does COD, and could be further

improved by using filtered samples to cut out interference from particulates.

Where waste contacts water, Phase III anaerobic acid formation is ongoing at the Paparua County Council Landfill. Although methane sampling was not undertaken, water sampling indicates conditions unsuitable for its formation. However, methane may be formed in anoxic pockets throughout the landfill and is known from landfill sites wherein groundwater does not contact waste (Knight 1984). Maturation is slow, since much of the waste lies above the watertable unless groundwater once again rises up into it but the effects of individual high rainstorm events inducing water percolation down through the landfill are unknown.

CHAPTER 7 CONCLUSIONS

CHAPTER 7 CONCLUSIONS

7.1 WASTE COMPOSITION

The Paparua County Council Landfill, originally a gravel pit sited in an old paleochannel of the Waimakariri River, was used as the main refuse disposal facility for municipal solid waste of Paparua County from early 1973 to July 1978. It is typical of many waste disposal sites established prior to 1980 which were poorly sited, poorly managed and simply filled existing depressions with no consideration of potential effects. Although records of refuse types were not kept, the site contents are assumed to be similar to presently disposed refuse. New Zealand is currently rated fifth largest producer of municipal solid waste per capita in the world. However, much of the waste is putrescible organic matter and lacks high quantities and concentrations of industrial waste. This indicates that along with inorganic constituents, leachates produced from New Zealand landfills will be dominated by naturally produced organics resulting from the degradation of organic matter rather than synthetic organic chemicals.

7.2 LEACHATE GENERATION

A Water Balance using averaged monthly rainfall totals, suggests that rainfall in Canterbury is too low to produce leachate. However high intensity rainfalls events and irrigation of the Greyhound Track, during the summer months, have an additional (undetermined) effect on the quantity of percolation through the P.C.C. Landfill surface. Newspaper reports and water level records, from well M36/890, illustrate that waste was periodically deposited directly into water, and on one occasion this included offal and sheep carcasses. Water also rose into previously dry refuse during the landfill operation, immediately producing leachate.

As a wet landfill the site rates as a high pollution potential with a DRASTIC Index of 187, and an unacceptable probable contamination risk, as a +14G or F site, under the Le Grand landfill classification. Such 'feasibility level' landfill evaluations allow for changes in site conditions due to engineering influences (e.g. liners) but take no account of the geometry and permeabilities of the local geologic setting nor the climatic or anthropogenic influences on moisture sources. In the Canterbury multilayer aquifer setting, no distinction is made between a site immediately underlain by a silt aquitard and one underlain by thick gravels, because both are so far above

basement. Therefore the pollution potential of each landfill must be individually assessed by a site-specific investigation which should include the history of waste deposition, and the evaluation of sediment permeability and geometry immediately around the site.

7.3 HYDROGEOLOGY AND GEOLOGY

Three wells installed at each of the two investigation sites, penetrated gravels and sandy gravels with transmissivities of 4622 m^2/day at Wigram and 27260 m^2/day at the P.C.C. Landfill. Small pods of clays were rarely encountered. At Wigram a thin laterally restricted aquitard is present at a depth of approximately 14 m in well M35/6102 . In contrast, the P.C.C. Landfill was underlain by an extensive silty sand varying in thickness but encountered at monitoring wells throughout the site including the BORE, SHEEP and DOG locations. The landfill base contacted the silty sand at a depth of approximately 5.9 m at the BORE location whilst at DOG a thin sandy gravel was intermediate between the two units.

Kanuka and Totara wood sampled at a depth of approximately 13 m within the silty sand unit at the P.C.C. Landfill site produced a radiocarbon age of 8820 ± 70 years B.P. and is taken to represent the depositional age of the unit. Similar wood types found at depth of approximately 14 m at Wigram produced a radiocarbon age of 6660 ± 60 years. Landward marine incursions during the Holocene produced swampy conditions in the Wigram / Halswell locality with finer sediments deposited in the usually braided fluvial setting. The variation in radiocarbon ages between the sites confirms the complicated depositional history of sedimentary deposits in the area.

The silty sand underlying the P.C.C. Landfill has a significantly lower permeability (2.9×10^{-9} to 3.2×10^{-7} m/day) than the deeper gravels and sands. This unit acts as a aquitard, basally confining the landfill and producing a slightly perched water table which is distinct from groundwater in the deeper gravel aquifer with a regional potentiometric gradient of 8.9×10^{-4} m/m.

Both international groundwater tracer studies and transport theory reported in the literature, have firmly swung groundwater contamination investigations away from the characterisation of vertically-averaged aquifer parameters to vertically-distributed properties. It is now widely recognised that full aquifer dispersivity has a much smaller

influence on contaminant transport than hydraulic conductivity and transport models would benefit from increased vertical and lateral aquifer property characterisation. In this regard, the point dilution technique has proven to be a low cost method of characterising preferred migration paths on the basis of groundwater velocities. Combining this technique with tracer tests is a logical step in evaluating groundwater behaviour and significantly improves the information gathered from a single test as exemplified by down-hole conductivity probe information in the Wigram tracer test.

At the P.C.C. Landfill, point dilution tests using low cost easily operated equipment, designed and built for the investigation, were undertaken in the gravels and sands beneath the silty sand aquitard. Laterally continuous high seepage velocity zones between two wells 10 m apart were identified. These may act as preferred contaminant transport paths for leachate percolating downward through the silty sand.

Despite the acknowledged problems of relating hydraulic properties to the physical properties of coarse and fine sediments, the velocity trends found using the point dilution technique, showed a strong correlation with size analyses of sediment taken from the boreholes. Hydraulic conductivities ranging from 4.4×10^{-2} to 9.2×10^{-1} m/s were obtained from 0.5 m packed off intervals beneath the silty sand aquitard unit in wells M36/4102 and M36/4017. Averaged hydraulic conductivities for each well were 1.32×10^{-4} and 7.14×10^{-4} m/s respectively. Given the errors involved, this was similar to the 2.33×10^{-3} m/s obtained by a standard constant discharge pump test at the site.

Down-hole geophysical sondes produced relatively featureless well logs at both the Wigram and P.C.C. Landfill sites, although they did locate the water table position and weld positions in steel cased wells. The presence of a leachate contaminant zone beneath the P.C.C. Landfill was not detected by down-hole nuclear logging in well M36/4107.

7.4 SURFACE GEOPHYSICS

The successful detection of leachate by electrical resistivity profiling at the only New Zealand trials to date, at the P.C.C Landfill and at the Kaiapoi Landfill (Broadbent 1991), proves the methods' application as a relatively low cost means of quickly ascertaining the presence and migration of contaminants. Contaminant detection by resistivity profiling is particularly suited to the irregular nature of the Canterbury

sediments. The spatial coverage of resistivity profiling can detect contaminant migration patterns where hydrochemical point sampling would be misleading. This is because the actual leachate migration direction may not be the same as the predicted downgradient direction due to the geometry of less permeable lenses or the non-aqueous behaviour of the contaminants, as illustrated in the tracer test at Wigram. The use of resistivity soundings to determine landfill stratigraphy was successfully illustrated for the first time in New Zealand in the P.C.C. Landfill investigation.

Surface geophysical techniques may prove to be the most cost effective means of ascertaining the presence and migration of leachate from Canterbury landfills, and assist in siting hydrochemical sampling points. It is stressed that each site has unique interference factors and surface geophysical investigations should follow the stepwise philosophy outlined in this investigation (e.g., with profiling only following successful soundings, and extensive profiling following the success of trial profile lines). Electromagnetic techniques may prove even more useful in the Canterbury setting, with the portable EM-34 eliminating the need for set-up and survey time.

The presence of saturated refuse was confirmed at the BORE and DOG locations of the P.C.C. Landfill where the best-fit models of both Wenner and Schlumberger soundings incorporated a low resistivity layer of 2.3 to 6.7 Ωm compared to values of 19.6 to 30.1 Ωm for unsaturated refuse, and >300 Ωm representing uncontaminated saturated gravels. The southern SHEEP locality was leachate-free but the near surface silty sand produced a 77 Ωm layer on the sounding model. Water levels were poorly defined by soundings.

The positions of electrodes in a stacked Wenner profiling array covering the full P.C.C. Landfill site were located by surveying prior to field measurements. Based on an 'a' spacing of 15 m, the shallower profile with a depth of investigation characteristic coincident with the water table (circa 5 m), revealed a low resistivity zone with values down to 14.4 Ωm which was constrained to the deepest extraction zone in the northeastern corner of the landfill. This zone represented the position of leachate. The deeper penetrating profile based on an 'a' spacing of 30 m, showed low apparent resistivity values extending to the southeast but values increased and flattened out with depth. The geometry of this low apparent resistivity zone indicates that leachate is constrained with depth. The position of low resistivity zones dictated the siting of

monitoring wells for hydrochemical analysis of groundwater.

In a seismic refraction traverse at the SHEEP location it was noted that the critical ray was not refracted where the site had been refuse infilled suggesting the method may prove useful in defining landfill boundaries.

The combination of surface resistivity and down-hole conductivity probes worked well in the detection of groundwater behaviour using an induced conductive contaminant at Wigram, but the technique is more suited to shallow aquifers where the water table is < 7 m deep. The inhomogeneous braided nature of the aquifer sediments was illustrated by the irregular pattern of the contaminant plume traced over an 8 day period. Expected resistivity decreases of 15 to 30% were not obtained due to tracer dilution and sinking. Retrospective one dimensional modelling of geoelectric soundings using tracer resistivities from down-hole conductivity probe measurements indicated a 9% decrease which was much closer to the 6% actually achieved in the field test. Seepage velocities between 3.58 and 8.52 m/day were similar to those found in gravelly sediments underlying the P.C.C. Landfill.

7.5 HYDROGEOCHEMISTRY

7.5.1 INORGANIC CONTAMINANTS

Refuse and sediment collected from logged wells at the P.C.C. Landfill were leached to assess the effects of ion desorption on refuse and paper in different solutes. A buffered acetic acid solvent, representing conditions at the base of a landfill, preferentially removed Ca, Mg, Fe and Mn ions from both mediums, with paper desorbing more ions per unit weight than refuse. Whilst the release of Ca, Mg, and $N\text{-}NO_3$ ions from paper is enhanced in the organic solvent as opposed to distilled water, SO_4, Cl and Br anion desorption was inhibited. The release of this latter anion group, along with Ca, were also inhibited in refuse. X-ray fluorescence analysis of paper showed that following fluid extraction, more Ca, Na and S ions were retained by paper derived from below the water table as opposed to paper higher in the waste pile but more of these ions were also released into the extractant fluid. The reverse applied for K. Thus the ability of paper to retain major ions and prevent their movement into groundwater may be transient. Both refuse and paper tend to retain the trace metals. This has been observed at a number of municipal solid waste landfills where trace metals are not detected at significant levels in the leachate

produced.

Trilinear plots showed a strong horizontal mixing regime with extractant end members derived from sediment; Ca + Mg and CO_3 and HCO_3 rich, and from refuse; Na + K and SO_4 + Cl rich. Groundwater samples all plotted within the mixing zone, although there was a subtle vertical trend from higher to lower reducing conditions as represented by the nitrogen index of samples.

Groundwater samples fell into two distinct groups. New Zealand drinking water quality guidelines were exceeded by Cu (0.11 mg/l), Cl (270 mg/l), Na (210 mg/l) and B (2.8 mg/l) in Group 1 samples, which also exhibited high levels of Ca, Mg, conductivity and COD. Ammoniacal-nitrogen levels up to 160 mg/l and TDS ranging from 1238 to 1736 mg/l were found in these samples. Group 1 samples were derived from wells which tapped the active leachate development zone. Although Fe and Mg were found at high levels in Group 2 samples, few other contaminants were found. Ammoniacal-nitrogen was present at concentrations < 10 mg/l and TDS values were <660 mg/l. Group 2 samples originated from wells either below the silty sand aquitard, or outside the northeast corner of the landfill. The high organic matter content of the refuse may assist in the attenuation of some of the contaminants noted from groundwater samples.

Total organic carbon values distinguished leachate-derived groundwater, with values > 58 mg/l, despite the fact that samples were only centrifuged and not filtered to remove particulates. The presence of particulates in fluids can lead to high concentrations of trace metals, Fe and SO_4, which could be misidentified as coming from leachate. Combining extractant and fluid results on bivariate plots can identify these anomalous samples and distinguish contaminant sources where more than one is suspected.

The presently used field-detectable groundwater quality parameters, Eh, pH, DO, temperature and conductivity, are often relatively blind to the presence of leachate contamination. Calcium or Cl ion concentrations increase at the leading edge of a leachate plume in a hardness halo, and B ions are indicative of a synthetic contaminant source. Thus a field useable ion specific electrode, responding to any one of these elements, may be useful in delineating contaminants without the time and cost

involved in laboratory analysis.

7.5.2 ORGANIC CONTAMINANTS

The P.C.C. Landfill is free from groundwater quality degradation by organic substances except in the immediate leachate development zone in the northeast corner of the site. Phthalate esters, found commonly in leachates from more recently completed or operating landfills, were absent due to the closure of the P.C.C. Landfill prior to the widespread use of plastic rubbish bags and packaging. Ketones, amines and terpenes account for the pungent odour and yellow discoloration of leachate samples, and derive from the degradation of putrescible organic waste. Xylenes, found at a concentration of 5 mg/m^3 in groundwater sampled from well M36/4317, alkyl benzenes, and S- and N- heterocyclic compounds, were the only synthetic organics found. Consequently, the landfill site is not seen as presently contributing substantial concentrations or types of organic contaminants to the underlying groundwater system. Trichloroethene, 1,1,1 trichloroethane, tetrachloroethene and trichloromethane found downgradient of the P.C.C. Landfill in groundwater surveys by the Canterbury Regional Council, were not found in leachate from the site.

Leachate characteristics indicate the landfill to be at an anaerobic acid-formation phase of stabilisation although this may vary widely throughout sections of the landfill.

7.6 IMPACT ON THE GROUNDWATER RESOURCE

7.6.1 REGIONAL

The information presented in this thesis is relevant to a short term hydrogeologic time period i.e., it is a mapping rather than a monitoring investigation, and has only established the presence of leachate at the P.C.C. Landfill site, without fully examining the generation factors. Such factors include climate change such as high intensity rainfall events leading to erratic percolation and leachate generation cycles, hydrogeological changes i.e., water levels rising higher into the waste pile as has occurred in the past, or chemical changes i.e., the release of substances initially adsorbed by sediment particles. There is no guarantee that leachate forming conditions will remain at a constant rate and strength; only long-term monitoring will address these issues.

The classic commonly cited example of a landfill leachate plume is that of the

Canadian Forces Base Borden site near Alliston, Ontario, which has been used for a variety of both induced and existent contaminant monitoring experiments (MacFarlane *et al.* 1983, Greenhouse *et al.* 1983, Nicholson *et al.* 1983, Dance & Reardon 1983, Sudicky *et al.* 1983, Sudicky 1986, Egboka *et al.* 1983, Cherry *et al.* 1983, Roberts 1986, Curtis 1986, Freyberg 1986). The step-shaped mounded landfill is of a similar size (5.4 ha.) and depth (5 to 10 m) to that of the P.C.C. Landfill. It is located above an unconfined fine to medium glaciofluvial sand with sandy and clayey interlayers and a hydraulic conductivity of 6×10^{-3} to 1×10^{-2} cm/s (5.2×10^{0} to 8.6×10^{0} m/day) (MacFarlane *et al.* 1983). This unit in turn overlies a silty-clay aquitard at a depth of 15 to 30 m, which confines the top of a lower aquifer used for groundwater extraction. The Borden landfill received putrescible refuse in the form of mess hall waste from the Forces Base in addition to construction debris, and is therefore regarded as a relatively clean landfill (Cherry 1981). Although most refuse is only 1 to 2 m above the water table throughout most of the year, water table changes may mean that an unsaturated sand intervening between the refuse base and the water table is nonexistent (Nicholson *et al.* 1983). The site therefore shares important features with the P.C.C. Landfill but unlike the latter is carefully monitored with over 5000 depth-integrated sampling points established.

Hydrochemical sampling and surface geophysics at the Borden site have shown the existence of a 700 m long, 600 m wide and 60 m deep leachate plume where chemical constituent levels exceed background. As at the P.C.C. Landfill, contaminated water was found to possess a pungent odour and yellowish discoloration. Whilst the contamination at Borden is extensive, S and Fe are the only inorganic constituents to exceed the recommended drinking water standards (Nicholson *et al.* 1983), whilst Fe, Mn and Zn are the only metals at detectable levels.

Two major conclusions can be made from a comparison of the Borden Landfill to the P.C.C. Landfill. Firstly, at the Borden site the leachate plume is defined on the basis of groundwater constituents exceeding background as characterised by the hydrochemistry of the surrounding aquifer. The geophysically-defined plume shows the major electrolyte area, whilst hydrochemistry invariably outlines a more extensive zone. The same effect is found in many contaminant studies, including the New Zealand example of Moore (1990). At the P.C.C. Landfill site 'background' has been taken in the geophysical survey where resistivity is typical of unextracted gravel

deposits. In the hydrochemical survey 'background' is taken to be the low chemical constituent levels, normally found in uncontaminated groundwater (Talbot *et al.* 1986). Because of the low number of monitoring wells used and the fact that the geophysically-defined plume is probably less extensive than the hydrochemically-defined plume, the full extent of the leachate at the P.C.C. Landfill has not been defined. It is possible that even the low levels of constituents found in groundwater beneath the landfill constitute leachate contamination *sensu stricto*. More precise background values should therefore be statistically defined for the Canterbury region.

Secondly, at the Borden site there is no evidence that the rate or strength of the leachate is declining appreciably with age. If this applies to landfills in Canterbury, abstraction of groundwater, for supply of drinking water, from the vicinity of each site must be prevented. The aquifers contain little organic matter or clays to adsorb contaminants and the greywacke gravels are relatively inert and cannot buffer the chemical effects of the leachate as would a carbonate system. Therefore until the presence or absence of leachate at each waste disposal site is established, compliance zones should be defined within which the diluting and dispersing nature of the aquifers have returned contaminants to background concentrations. This should include hardfill tips for which content regulation is difficult. For example, drinking water wells should not be installed within 100 m of the boundary to any waste disposal site and wells up to 500 m downgradient should abstract water from a depth in excess of 15 m.

7.6.2 PAPARUA COUNTY COUNCIL LANDFILL

Despite local groundwater contamination, the P.C.C. Landfill is considered a low risk locality in terms of leachate impact on drinking water abstracted from the underlying aquifer. Leachate is presently laterally restricted to only the most deeply extracted northeast section of the P.C.C. Landfill. Leachate strength rapidly attenuates with depth and is inhibited from deeper migration due to the presence of an underlying silty sand unit. It is possible that intact containers of contaminants disposed of in the landfill have yet to release their contents, and may not do so for many years. However, given the basal confinement of the landfill, any disposed organics which do not become subject to microbial degradation, or adsorption, within the landfill reactor should be retained at the site for some time. Any soluble mobile leachate constituents which managed to permeate the silty sand would be rapidly diluted and carried away from the site through laterally continuous gravel and sand units.

Wells closest to the landfill abstract groundwater from beneath the silty sand aquitard and are presently unaffected by leachate. In addition, with the extension of housing northward from Halswell and southward from Wigram future groundwater abstraction from the locality will be minimised with the installation of water reticulation systems. These two factors mean that the risk of health problems from the groundwater is minimal because of the factors controlling leachate generation, composition and migration.

However, this presents a further paradox. Despite moving closer to a possible groundwater contaminant source, in this case an old landfill, the land occupants may be at little risk from contaminated groundwater but they may themselves induce groundwater contamination if the vulnerability of the groundwater resource in the western unconfined zone is not fully recognised and the usual practices of waste disposal, such as pouring oil directly onto the ground, where it is free to percolate downward to groundwater, is continued. The control of such waste disposal practices remains in the sphere of public education.

The P.C.C. Landfill is a paradoxical example of waste disposal practices and subsequent groundwater contamination by leachate. Given the operational history of the site and its setting there appears to be no worse scenario for contamination of the groundwater resource. However, hydrogeological, hydrogeochemical and geological investigations show it to be a relatively secure containment site where leachates presently constitute little risk to the quality of the underlying groundwater resource. However, the fact that this landfill presently appears relatively clean in terms of leachate development and migration cannot be used to indicate that others in the Canterbury district will behave similarly, and the P.C.C. Landfill should be considered a fortuitous case of leachate containment.

7.7 SUGGESTIONS FOR FURTHER RESEARCH

1. Whilst this investigation was instigated as a mapping project to give a "snapshot" of landfill behaviour, longer term monitoring, particularly of water levels, is recommended. Groundwater levels during this investigation were very low but water rises into refuse in the past, may occur again. In addition the relationship of groundwater levels in the vicinity of the Greyhound Track to the irrigation sprinklers, and whether some of the wells have continually perched or varying piezometric heads requires further investigation. The nature of the hydraulic communication between groundwater, beneath and above the silty sand unit, would be resolved by using time-series analysis of frequent waterlevel monitoring in wells located in these positions.

2. To the authors knowledge, no long term hydrochemical monitoring to establish a statistical basis of background groundwater quality and the temporal quality changes that occur in Christchurch aquifers, has been undertaken. The network of monitoring bores at the P.C.C. Landfill would be useable for this purpose.

3. Galvanic resistivity investigations at other New Zealand landfills should be based on the stepwise approach used at the P.C.C. Landfill. In addition, if and when electromagnetic and/or ground penetrating radar techniques become readily available in New Zealand, the P.C.C. Landfill would be an ideal site to compare the techniques.

4. Field studies of two- and three-dimensional aquifer heterogeneity characterisation are scarce and should be encouraged. However, as with the point dilutions undertaken at the P.C.C. Landfill, such work is time consuming and would benefit by correlation with other more easily calibrated techniques such as flowmeters.

5. Salt-water tracing experiments are not recommended in conditions where a basal unit is not clearly defined, and where the water table is as deep as 7 m. At Wigram, these conditions gave rise to insufficient resistivity contrast between virgin and contaminated ground to interpret the results in terms of hydraulic characteristics and the plume pattern may reflect a larger natural variation than accounted for by a 1% error. Future salt-water tracer studies should therefore more fully detail background variation and ensure the use of techniques such as downhole conductivity probes to confirm the tracer's behaviour.

6. Attempts to quantify full aquifer dispersivity in Canterbury, or elsewhere, should be discouraged on the basis that the concept is outdated and misleading.

ACKNOWLEDGEMENTS

To the many people who contributed to the learning process represented by this thesis, I would like to express my sincere appreciation.

Foremost of these are the Canterbury Regional Council, the University Grants Committee, and the former Paparua and Waimairi County Council's who funded this project. Both owners and leasers of the two investigation sites are thanked for making the land available and for co-operating with the investigation requirements and restrictions. These include the Paparua County Council, Peter Partridge and his flock, the Canterbury Owners and Breeders Club, the Canterbury K? t Club and the RNZAF Wigram Airforce Base personnel.

Staff of the Canterbury Regional Council are thanked for their technical support and guidance. In particular the contribution of Mike Freeman, who initiated, fostered and supervised the project is greatly appreciated. Brian Todd undertook much of the field work and taught me everything I know about pumps. John Weeber and John Hughey oiled the wheels of the permit application process, whilst Susan Timm, Catherine Moore, Ken Baxter, Peter Callander and Rod Dixon gave useful advice and discussion at various stages of this investigation.

I am indebted to the personnel of the DSIR Geology and Geophysics, Groundwater Group, in particular Paul White, Mike Broadbent and Mike Simpson for making available geophysics equipment and for enlightening discussions, technical assistance and advice. The analytical work undertaken by DSIR Water Chemistry Division, often at last minute notice, is also greatly appreciated.

The technical and academic staff of the Geology Department, University of Canterbury, are all thanked for their contribution. In particular Arthur Nicholas should be knighted for his ability to turn wild descriptions into working apparatus! Jim Cole's supervision, editing assistance and skill in holding the ends of the project together through its duration is gratefully acknowledged. Likewise the efforts of David Bell in initiating the project, and David Nobes who arrived on the scene in the final act and learnt me to write proper, are appreciated.

The discussion and support provided by Mike Noonan (Lincoln University) and Mark Milke (Civil Engineering Department, University of Canterbury) when the project once or twice hit thin ice was greatly appreciated. Mark is also thanked for his advice with Chapter 6. Thanks also to Dave MacPherson for instructing me in the use of the TOC analyser, and the McMillan drillers for working around my ravings.

Fellow students and flatmates have encouraged and supported the work in innumerable ways. Rose Fitzgerald has my heartfelt thanks for accepting the atmospheric oscillations induced in our work space with so much patience and humour. Joan Esterle and Todd Warner needed to be kick started with a coffee or three, but taught me a lot about perspectives. Nicky Deacon and Andrea Lobb provided inspiration despite the time zone difference. Helen Oliver fended off the fierce woolly beasts of the wild Wigram hinterland like no other jaffa could, while Craig Fergusson is the best leachate procurer in the west! Andy Nicol is thanked not only for his editing skills, but for his sustaining companionship and honesty in the marathon that was our Canterbury years. Jane Shearer is thanked for her stauch support in times of

system battling. To Catherine Moore I owe many things, not least of which is the beginning of this project and the inspiration provided by her own. Many precious hours of Cath's time was spent reviewing material and nurturing a fellow hydrohead's morale. These folk cannot be thanked enough for the dimensions they added to postgraduate life.

Encouragement, meals and the normality (read sometime madness) of life outside the department was at times provided by Bruce & Mette and the twins, Kez, Ruth, Craig, Carol, Erin, Sean, Cath, Marco, Julie, Rob, Kelvin, Lynne, Nathan and Hamish, for which I am indebted; and yes Hamish, this means I finally get to leave school after a zillion years and start my real education!

Finally, and most importantly, the support and encouragement of both my families who endured this saga is acknowledged. Special thanks to my partner Wayne and my mother Audrey, without whom I would neither have commenced, nor completed, this project, and of whom words fail to express my regard.

REFERENCES

REFERENCES

ACKWORTH, R.I., GRIFFITHS, D.H., 1985: Simple Data Processing of Tripotential Apparent Resistivity Measurements as an Aid to the Interpretation of Subsurface Structure. Geophysical Prospecting 33: 861-887.

ALAO, D.A., 1985: Locating Contaminants in Ground-Water System by Direct Current Resistivity Technique. Bulletin of the Association of Engineering Geologists XXII(1): 95-100.

ALEXANDER, M., LUSIGMAN, B.K., 1966: Effects of Chemical Structure on Microbial Degradation of Substituted Benzenes. Journal of Agricultural Food Chemistry 14: 410-413.

ALLER, L., BENNETT, T., LEHR, J.H., PETTY, R.J., 1985: DRASTIC: A Standardised System for Evaluation Ground Water Pollution Potential using Hydrogeologic Settings. Environmental Protection Agency, Washington D.C. EPA/600/2-85/018. 130p.

ALTON JAMESON, J., 1989: Sanitary Landfills and the Environment. In: Waste Management Institute New Zealand Inc. Conference, Nov. 1, Wellington, New Zealand.

BAEDECKER, M.H., BACK, W., 1979: Hydrogeological Processes and Chemical Reactions at a Landfill. Ground Water 17: 429-437.

BARCZEWSKI, B., MARSCHALL, P., 1989: The Influence of Sampling Methods on the Results of Groundwater Quality Measurements. In: Contaminant Transport in Groundwater, Kobus & Kinzelbach Eds., Balkema, Rotterdam, 33-39.

BARKER, R.D., 1979: Signal Contribution Sections and their Use in Resistivity Surveys. Geophysical Journal of the Royal Astronomical Society 59: 123-129.

BARKER, R.D., 1991: Improving the Quality of Resistivity Sounding Data in Landfill Studies. In: Geotechnical and Environmental Geophysics No. 5, Vol. II. Environment and Groundwater. Ed. by S.H. Ward. Society of Exploration Geophysicists, 245-251.

BARLAZ, M.A., HAM, R.K., SCHAEFER, D.M., 1990: Methane Production from Municipal Refuse: A Review of enhanced Techniques and Microbial Dynamics. Critical Reviews in Environmental Control 19(6): 557-584.

BATES, B.D., DAVIDGE, S., 1983: There's Salt in my Shadoof. Water & Soil 1: 7-10.

BELL, D.H., WILSON, D.D., 1988: Hazardous Waste Disposal in New Zealand. Bulletin of the International Association of Engineering Geology 37: 15-26.

BENNETT, D.B., BUSACCA, M., CLARK, E.E., 1988: The Modified Reverse-Circulation Air Rotary Drilling Technique for Development of Observation and

Monitoring Wells at Ground-Water Contamination Sites. In: Ground-Water Contamination: Field Methods, ASTM STP 963. A.G. Collins and A.I. Johnson Eds., American Society for Testing and Materials, Philadelphia, 146-161.

BENSON, R.C., 1989: Geophysical Techniques for Sensing Buried Wastes and Waste Migrations. Environmental Monitoring Systems Laboratory, Office of Research and Development, USEPA, Las Vegas, Nevada 89114. 236p.

BHATTACHARYA, P.K., PATRA, H.P., 1979: Direct Current Geoelectrical Sounding, Principles and Interpretation. Methods in Geochemistry and Geophysics Vol. 9. Elsevier.

BORDEN, R.C., YANOSCHAK, T.M., 1990: Ground and Surface Water Quality Impacts of North Carolina Landfills. Water Resources Bulletin 26(2) 269-277.

BOULAY, M.C., 1989: A New Political Geography of Hazardous Industrial Wastes: A Case Study of the Woolston Industrial Zone. Unpublished M.A. Thesis, Geography Department, University of Canterbury, New Zealand.

BOULAY, M.C., 1991: Christchurch Metropolitan Landfill Investigation, Review of Dispensation Conditions, Hazardous Waste Disposal Guidelines and Groundwater Quality Monitoring Programme. A report to the Canterbury Regional Council, Eco Source Ltd, Christchurch. 31p.

BOUWER, H., 1978: Groundwater Hydrology. McGraw Hill Book Co., New York, New York 10020. 480p.

BOWDEN, M.J., AYREY, R.B., 1981: Refuse Tips and Underground Water Quality, Waste Committee Meeting Report 30/1/81, North Canterbury Catchment and Regional Water Board. 6p.

BOWEN, R., 1986: Groundwater, 2nd Edition. Elsevier Applied Science Publication, Great Britain. 427p.

BROADBENT, M., 1978: Seismic Refraction Surveys for Canterbury Groundwater Research. Report No. 131, Geophysics Division, DSIR, New Zealand. 66p.

BROADBENT, M., 1984: Preliminary Interpretation of Electrical Resistivity Field Sounding Curves Using Two-layer Standard Curves. DSIR Internal Report, 4 April 1984.

BROADBENT, M., 1991: Leachate Detection using Electrical Resistivity Measurements at a Refuse Disposal Landfill near Kaiapoi. DSIR Geology and Geophysics. 34p.

BROWN, L.J., 1973: Sheet S76 - Kaiapoi. Geological Map of New Zealand, 1:63,360, Wellington, Department of Scientific and Industrial Research.

BROWN, L.J., WILSON, D.D., 1988: Stratigraphy of the Late Quaternary Deposits of the Northern Canterbury Plains, New Zealand. New Zealand Journal of Geology and Geophysics 31: 305-335.

BUCHAN, G., JARVIS, P., LEONARD, B., 1991: Towards Improved Waste Reduction and Recycling Programmes for Christchurch and Canterbury. Soil Science Department, Lincoln University, Canterbury, New Zealand. 34p.

BUCHAN, S., KEY, A., 1956: Pollution of Ground Water in Europe. Bulletin of the World Health Organisation 14: 949-1006. Geneva.

BURGES, S.J., 1986: Trends and Directions in Hydrology. Water Resources Research 22(9): 15-58.

CALKIN, S., 1989: A Shallow Seismic Refraction Survey of the Mallard North Landfill - Hanover Park, Illinois. Unpublished M.S. Thesis, Northern Illinois University.

CALLANDER, P.F., 1988: Aquifer Tests at Wigram Air Force Base, January 1988. North Canterbury Catchment Board Internal Report. 12p.

CAMPELLA, R.G., WEEMEES, I., 1990: Development and Use of an Electrical Resistivity Cone for Groundwater Contamination Studies. Canadian Geotechnical Journal 27: 557-567.

CARPENTER, P.J., CALKIN, S.F., KAUFMANN, R.S., 1991: Assessing the Fractured Landfill Cover Using Electrical Resistivity and Seismic Refraction Techniques. Geophysics 56(11): 1896-1904.

CARPENTER, E.W., HABBERJAM, G.M., 1956: A Tri-Potential Method of Resistivity Prospecting. Geophysics XXI(2): 455-469.

CARPENTER, P.J., KAUFMAN, R.S., PRICE, B., 1990: Use of Resistivity Soundings to Determine Landfill Structure. Ground Water 28(4): 569-575.

CARTWRIGHT, K., McCOMAS, M.R., 1968: Geophysical Surveys in the Vicinity of Sanitary Landfills in Northeastern Illinois. Ground Water 6(5): 23-30.

CAVALIER, T.C., LAVY, T.L., MATTICE, J.D., 1991: Persistence of Saturated Pesticides in Ground-Water Samples. Ground Water 29(2): 225-231.

CHEREMISINOFF, P.N., GIGLIELLO, K.A., O'NEILL, T.K., 1984: Groundwater Leachate: Modelling / Monitoring/ Sampling. Technomic Publishing Company Inc. Pennsylvania. 146p.

CHERRY, J.A., BARKER, J.F., BUSZKA, P.M., HEWETSON, J.P., MAYFIELD, C.I., 1981: Contaminant Occurrence in an Unconfined Sand Aquifer at a Municipal Landfill. In: Proceedings of the Fourth Annual Madison Conference of Applied Research and Practice on Municipal and Industrial Waste.

CHERRY, J.A., 1983: Occurrence and Migration of Contaminants in Groundwater at Municipal Landfills on Sand Aquifers. Chapter 13. In: Environment and Solid Waste - Characterisation, Treatment, and Disposal. Proceedings of the 4th Life Sciences Conference, ed. by C.W. Francis and S.I. Auerbach. Ann Arbor Science. 127-147p.

CHERRY, J.A., GILLHAM, R.W., ANDERSON, E.G., JOHNSON, P.E., 1983: Migration of Contaminants in Groundwater at a Landfill: A Case Study: 2. Groundwater Monitoring Devices. Journal of Hydrology 63: 31-49.

CLARK, T.P., PISKIN, R., 1977: Chemical Quality and Indicator Parameters for Monitoring Landfill Leachate in Illinois. Environmental Geology 1: 329-339.

CLOSE, M.E., 1991: Assessment of Christchurch Metropolitan Landfill groundwater quality data. Contract Report 1991/53 prepared for the Canterbury Regional Council. D.S.I.R. Geology and Geophysics 25p.

CORBITT, R.A., 1989: A Standard Handbook of Environmental Engineering. McGraw Hill Publishing Co., New York.

COCKS, J., EVANS, R., O'GRADY, J., ROWDEN, I., 1989: Regional Waste Management Plans - Practical Implications. In: Waste Management Institute New Zealand Inc. Conference, Nov. 1, Wellington, New Zealand.

COULTER, J.D., 1973: A Water Balance Assessment of the New Zealand Rainfall. Journal of Hydrology (NZ) 12(2): 83-91.

COX, J.E., MEAD, C.B., 1978: Soils and Agriculture of Part Paparua County. Soil Bureau Bulletin 34. Soil Bureau, DSIR, Wellington. 128p.

COX, J.E., MEAD, C.B., 1978: New Zealand Soil Bureau Map 1: 31,680.

CURTIS, G.P., ROBERTS, P.V., REINHARD, M., 1986: A Natural Gradient Experiment on Solute Transport in a Sand Aquifer. 4. Sorption of Organic Solutes and its Influence on Mobility. Water Resources Research 22(13): 2059-2067.

DANCE, J.T., REARDON, E.J., 1983: Migration of Contaminants in Groundwater at a Landfill: A Case Study. 5. Cation Migration in the Dispersion Test. Journal of Hydrology 63: 109-130.

DAVIS, S.N., DeWIEST, R.J.M., 1966: Hydrogeology. John Wiley & Sons Inc., New York. 463p.

DAVIS, S.N., THOMPSON, G.M., BENTLEY, H.W., STILES, G., 1980: Ground-Water Tracers - A Short Review. Ground Water 18(1): 14-23.

DAVIS, S.N. et al 1985: An Introduction to Ground-Water Tracers. EPA/600/2-85/022, PB 86-100591, NTIS, Springfield, Virginia.

DAVISON, A.S., 1969: The Effect of Tipped Domestic Refuse on Groundwater Quality. Water Treat. Exam. 18(1): 35-41.

DEPARTMENT OF HEALTH 1984: Drinking-Water Standards for New Zealand. A Report Prepared for the Board of Health, Wellington. 49p.

DOBRIN, M.B., 1976: Introduction to Geophysical Prospecting, 3rd Edition. McGraw Hill, New York.

DOUGLAS, M., 1977: Chapter 13: General Planning and Other Factors. In: Christchurch Metropolitan Refuse Committee. Disposal for Christchurch Including the Waimairi Coastal and Recreation Plan. Environmental Impact Report Vol. 1 & 2. 14p.

DRISCOLL, F.G., 1986: Groundwater and Wells, 2nd Edition. Johnson Division, St. Paul, Minnesota 55112, 1089p.

DROST, W., NEUMAIER, F., 1974: The application of single-borehole methods in groundwater research. International Atomic Energy Agency, Symposium on Isotope Techniques in Groundwater Hydrology, Vienna, IAEA-SM, 182-912.

EBRAHEEM, A.M., HAMBURGER, M.W., BAYLESS, E.R., KROTHE, N.C., 1990: A Study of Acid Mine Drainage using Earth Resistivity Measurements. Ground Water 28(3): 361-368.

EDWARDS, L.S., 1977: A Modified Pseudosection for Resistivity and Induced Polarisation. Geophysics 42: 1020-1036.

EGBOKA, B.C.E., CHERRY, J.A., FARVOLDEN, R.N., FRIND, E.O., 1983: Migration of Contaminants in Groundwater at a Landfill: A Case Study. 3. Tritium as an Indicator of Dispersion and Recharge. Journal of Hydrology 63: 51-80.

EGBOKA, B.C.E., UMA, K.O., 1986: Comparative Analysis of Transmissivity and Hydraulic Conductivity Values from the Ajali Aquifer System of Nigeria. Journal of Hydrology 83: 185-196.

ELLIS, D.V., 1987: Well Logging for Earth Scientists. Elsevier, New York. 532p.

EVJEN, H.M., 1938: Depth Factor and Resolving Power of Electrical Measurements. Geophysics 3: 78-95.

FABBRI, L.A., RODDEN,P., 1990: Conducting Comprehensive Environmental Investigations of Solid Waste Landfills in Massachusetts. In: Water Quality Management of Landfills, Water Pollution Control Federation Specialty Conference Series, Chicago, Illinois. 2.1-2.18.

FARVOLDEN, R.N., HUGHES, G.M., 1976: Hydrogeological Implications in Solid Waste Disposal. Bull. Inter. Assoc. Sci. Hydrol.: 146-158.

FENN, D.G., HANLEY, K.J., DeGEARE, T.V., 1975: Use of the Water Balance Method for Predicting Leachate Generation from Solid Waste Disposal Sites. EPA Washington D.C., EPA SW-168. 40p.

FETTER, C.W., 1988: Applied Hydrogeology, 2nd Edition. Merrill Publishing Company, Columbus, Ohio. 592p.

FOLK, R.LO., ANDREWS, P.B., LEWIS, D.W., 1970: Detrital Sedimentary Rock Classification and Nomenclature for use in New Zealand. New Zealand Journal of Geology and Geophysics 13: 937-968.

FRANKLIN ASSOC. INC., 1988: Characterization of Municipal Solid Waste in the United States 1960 to 2000 (Update of 1986 version). EPA/530/SW-88/033. 43p.

FREEMAN, M., 1989a: Report on a Survey of Groundwater and Surface Waters for Organic Contaminants. Unpublished Canterbury Regional Council Internal Report.

FREEMAN, M., 1989b: Results of Groundwater Quality Monitoring for Chlorinated Solvents in the Hillmorton area. Unpublished Canterbury Regional Council Internal Report.

FREEZE, R.A., CHERRY, J.A., 1979: Groundwater. Prentice Hall Inc., New Jersey. 604p.

FREYBERG, D.L., 1986: A Natural Gradient Experiment on Solute Transport in a Sand Aquifer. 2. Spatial Moments and the Advection and Dispersion of Nonreactive Tracers. Water Resources Research 22(13): 2031-2046.

FRIED, J.J., 1975: Groundwater Pollution, Theory, Methodology, Modelling and Practical Rules. Elsevier Scientific Publishing Company, Amsterdam. 330p.

FROLICH, R.K., PARKE, C.D., 1989: The Electrical Resistivity of the Vadose Zone - Field Survey. Ground Water 27(4): 524-530.

FUNGAROLI, A.A., 1971: Pollution of Subsurface Waters by Sanitary Landfills. United States Environmental Protection Agency Publication SW-12. 131p.

FURTAK, H., LANGGUTH, H.R., 1967: Zur hydrochemischen keinnieichnung von grundwasern und grundwassertypen mittels kennzahlen. Int. Assoc. Hydrogeol. Mem. 7: 89-96. Hannover.

GASPAR, E., ONCESCU, M., 1972: Radioactive Tracers in Hydrology. Elsevier Publishing Company, New York. 342p.

GEBHARDT, K., JANKOWSKI, J., 1987: Preliminary Landfill Siting and Related Analysis using Simple Modeling Techniques. Engineering Geology 23: 291-306.

GELHAR, L.W., MANTOGLOU, C., WELTY, C., REHFELT, K.R., 1985: A Review of Field Scale Physical Transport Processes in Saturated and Unsaturated Porous Media. Report RP-2485-05, Electric Power Research Institute, Palo Alto, California.

GEOLOGICAL SOCIETY ENGINEERING GROUP WORKING PARTY,(G.S.E.G.), 1988: Engineering Geophysics Report. Quarterly Journal of Engineering Geology. London. 21: 207-271.

GHOSH, D.P., 1971: Inverse Filter Coefficient for the Computation of Apparent Resistivity Standard Curves for a Horizontally Stratified Earth. Geophysical Prospecting 19: 769-775.

GREENHOUSE, J.P., HARRIS, R.D., 1983: Migration of Contaminants in

Groundwater at a Landfill: A Case Study. 7. DC and VLF and Inductive Resistivity Surveys. Journal of Hydrology 63: 177-197.

GREENHOUSE, J.P., MONIER-WILLIAMS, M., 1985: Geophysical Monitoring of Ground Water Contamination around Waste Disposal Sites. Ground Water Monitoring Review 5(4): 63-69.

GREENHOUSE, J.P., SLAINE, D.D., 1986: Geophysical Modelling and Mapping of Contaminated Groundwater around Three Waste Disposal Sites in Southern Ontario. Canadian Geotechnical Journal 23: 372-384.

GRIFFITHS, D.H., KING, R.F., 1969: Applied Geophysics for Geologists and Engineers: Elements of Geophysical Prospecting. Pergammon Press Ltd. 230p.

GRISAK, G.E., MERRITT, W.F., WILLIAMS, D.W., 1977: A Fluoride Borehole Dilution Apparatus for Groundwater Velocity Measurements. Canadian Geotechnical Journal 14: 554-561.

GUNN, A., 1983: Hazardous Wastes. Soil & Water 19(2): 4-9.

GUTHRIE, M., 1986: Use of a Geo Flowmeter for the Determination of Ground Water Flow Direction. Ground Water Monitoring Review 6(1): 81-86.

HABBERJAM, G.M., WATKINS, G.E., 1985: The Reduction of Lateral Effects in Resistivity Probing. Geophysical Prospecting XV(2): 221-235.

HACKBARTH, D.A., 1971: Field Study of Subsurface Spent Sulfite Liquor Movement Using Earth Resistivity Measurements. Ground Water 9(3): 11-16.

HADLEY, P.W., ARMSTRONG, R., 1991: "Where's the Benzene?" - Examining California Ground Water Quality Surveys. Ground Water 29(1): 35-40.

HALEVY, E., MOSER, H., ZELLHOFER, O., ZUBER, A., 1967: Borehole Dilution Techniques: A Critical Review. Isotopes in Hydrology. International Atomic Energy Agency, Vienna, 531-564.

HAM, R.K., 1975: The Generation, Movement and Attenuation of Leachate from Solid Waste Land Disposal Sites. Waste Age 6(6): 50-59 and 111-112.

HAM, R.K., 1976: Solid Waste Degradation Due to Shredding and Sludge Addition. In: Gas and Leachate from Landfills; Formation, Collection and Treatment. Rutgers. EPA 600/9-76-004. E,J. Genetelli and J. Cirello Eds., NTIS PB 251-161. R168-176.

HARRIS, R.C., LOW, D.R., 1984: Changes in the Organic Fraction of Leachate from Two Domestic Refuse Sites on the Sherwood Sandstone, Nottinghamshire. Quarterly Journal of Engineering Geology, London 17: 57-69.

HAZEN, A., 1893: Some Physical Properties of Sands and Gravels. Massachusetts State Board of Health, 24th Annual Report.

HEM, J.D., 1985: Study and Interpretation of the Chemical Characteristics of Natural

Water. 3rd Edition. U.S. Geol. Surv. Water Supply Paper 2254, 263p.

HENSEL, B.R., KEEFER, D.A., GRIFFIN, R.A., BERG, R.C., 1991: Numerical Assessment of a Landfill Compliance Limit. Ground Water 29(2): 218-224.

HERZER, R.H., 1981: Late Quaternary Stratigraphy and Sedimentation of the Canterbury Continental Shelf, New Zealand. New Zealand Oceanographic Memoir 89. Department of Scientific and Industrial Research.

HESS, K.M., 1989: Use of a Borehole Flowmeter to Determine Spatial Heterogeneity of Hydraulic Conductivity and Macrodispersion in a Sand and Gravel Aquifer, Cape Cod, Massachusetts. In: Proceedings of the Conference on New Field Techniques for Quantifying the Physical and Chemical Properties of Heterogeneous Aquifers, Dallas, Texas. F.J. Molz, J.G. Melville and O. Guven Eds., National Water Well Association, Water Well Journal Publishing Co., Ohio. 497-507.

HICKMAN, H.L., 1990: Sanitary Landfills - What's the Problem? In: Water Quality Management of Landfills. Water Pollution Control Federation Specialty Conference Series, Chicago, Illinois. 1.1-1.8p.

HOEKS, J., BORST, R.J., 1982: Anaerobic Digestion of Free Volatile Fatty Acids in Soils Below Waste Tips. Water, Air and Soil Pollution 17: 165-173.

HOEKSTRA, B., HOEKSTRA, P., 1990: Planning and Executing Geophysical Surveys. In: Proceedings of the Fourth National Outdoor Action Conference on Aquifer Restoration, Ground Water Monitoring and Geophysical Methods, Las Vegas, Nevada. 1159-1166p.

HOLMES, R., 1984: Comparison of Different Methods of Estimating Infiltration at a Landfill Site in South Essex with Implications for Leachate Management and Control. Quarterly Journal of Engineering Geology, London 17: 9-18.

JAYET, D.F., 1986: An Examination of Observed Climatic Trends / Changes Over Banks Peninsula and the Surrounding Plains Area, and their Synoptic Climatology. Unpublished M.Sc. Thesis, University of Canterbury, New Zealand.

JENSEN, M.L., BATEMAN, A.M., 1981: Economic Mineral Deposits. Third Edition, Revised Printing. John Wiley & Sons, New York. 593p.

KEELY, J.F., BOATENG, K., 1987: Monitoring Well Installation, Purging, and Sampling Techniques - Part 1: Conceptualizations. Ground Water 25(3): 300-313.

KENLEY, P.R., 1979: Landfill Methods Suitable for the Disposal of Intractable Wastes. Interdepartmental Committee on Liquid Waste Disposal, Unpublished Report. 9p.

KENOYER, G., SEAGERG, J., REESE, J., HESS, G., BURTON, G.A., 1990: Simulation of Aquifer Remediation with Laboratory and Field Tests of Sorption of Chlorinated VOCs. In: Proceedings of the Fourth National Outdoor Action

Conference on Aquifer Restoration, Ground Water Monitoring and Geophysical Methods, Las Vegas, Nevada. 781-795p.

KERFOOT, W.B., 1988: Monitoring Well Construction and Recommended Procedures for Direct Ground-Water Flow Measurements using a Heat-Pulsing Flowmeter. In: Ground-Water Contamination: Field Methods, ASTM STP 963. A.G. Collins and A.I. Johnson Eds., American Society for Testing and Materials, Philadelphia, 146-161.

KERFOOT, W.B., KIELY, L., 1989: A Direct-Reading Borehole Flowmeter. In: Proceedings of the Conference on New Field Techniques for Quantifying the Physical and Chemical Properties of Heterogeneous Aquifers, Dallas, Texas. F.J. Molz, J.G. Melville and O. Guven Eds., National Water Well Association, Water Well Journal Publishing Co., Ohio. 579 - 584.

KIDD, C.H., HANCOCK, J.S., 1983: Applicability of Le Grand's Standardised System for Evaluating Waste Disposal Sites in Australia and New Zealand. In: International Conference on Groundwater and Man, Sydney, Vol. 2. Australian Water Resources Council Conference Series No.8. Department of Resources and Energy, Australian Government Publishing Service, Canberra. 207-218p.

KILLEY, R.W.D., MOLTYANER, G.L., 1988: Twin Lake Tracer Tests: Setting, Methodology, and Hydraulic Conductivity Distribution. Water Resources Research 24(10): 1585-1612.

KIMMEL, G.E., BRAIDS, O.C., 1974: Leachate Plumes in a Highly Permeable Aquifer. Ground Water 12(6): 388-393.

KING, G., RUMBLE, B., SIMPSON, J., 1989: New Zealand's First Landfill Gas Control System. In: Waste Management Institute New Zealand Inc. Conference, Nov. 1, Wellington, New Zealand.

KLEFSTAD, G., SENDLEIN, L.V.A., PALMQUIST, R.C., 1975: Limitations of the Electrical Resistivity Method in Landfill Investigations. Ground Water 13(5) 418-427.

KNIGHT, M.J., 1984a: Geological Factors in Siting Landfills and New Fill Design Concepts. In: Landfill Disposal of Wastes and Effects on Groundwater and Streams. Knight, M.J. (Ed.) 1.1-1.22p.

KNIGHT, M.J., 1984b: Chemical and Biological Processes Associated with Landfills for Industrial Wastes. In: Landfill Disposal of Wastes and Effects on Groundwater and Streams. Ed. Knight, M.J. 22p.

KOEFOED, O., 1979: Resistivity Sounding Methods. Methods in Geochemistry and Geophysics, Elsevier.

KRUSEMAN, G.P., De RIDDER, N.A., 1976: Analysis and Evaluation of Pumping Test Data. International Institute for Land Reclamation and Improvement, Wagenigen, Netherlands. 200p.

LAMB, A.J.W., FORBES, B.D., 1977: Location and Management of Landfill Areas.

Chapter 3: Refuse Disposal. In: Christchurch Metropolitan Refuse Committee. Disposal for Christchurch Including the Waimairi Coastal and Recreation Plan. Environmental Impact Report Vol. 2.

LE GRAND, H.E., 1964: System for Evaluation of Contamination Potential of Some Waste Disposal Sites. Journal of American Water Works Association 56: 957-974.

LE GRAND, H.E., 1983: A Standardised System for Evaluating Waste-Disposal Sites. National Water Well Association, Worthington, Ohio. 49p.

LEE, G.F., JONES, R.A., 1991: Guest editorial - Landfills and ground-water quality. Ground Water 29(4): 482-486.

LEHR, J.H., 1988: An Irreverent View of Contaminant Dispersion. Ground Water Monitoring Review 8(4): 4-6.

LEMA, J.M., MENDEZ, R., BLAZQUEZ, R., 1988: Characteristics of Landfill Leachates and Alternatives for their Treatment: A Review. Water, Soil and Air Pollution 40: 223-250.

LITTLE, C., 1990: Recycled Oils: The Perfect Solution. Newsletter of the Waste Management Institute of New Zealand. Waste Observer Issue 5: 33-35.

LU, J.C.S., EICHENBERGER, B., STEARNS, R.J., 1985: Leachate from Municipal Landfills: Production and Management. NOYES Publications, New Jersey. 453p.

MacFARLANE, D.S., CHERRY, J.A., GILLHAM. R.W., SUDICKY, E.A., 1983: Migration of Contaminants in Groundwater at a Landfill: A Case Study. 1. Groundwater Flow and Plume Delineation. Journal of Hydrology 63:1-30.

McGINLEY, P.M., KMET, P., 1984: Formation, Characteristics, Treatment and Disposal of Leachate from Municipal Solid Waste Landfills. A Wisconsin Department of Natural Resources Special Report. Bureau of Solid Waste Management. 288p.

MacLEAN, R.D., 1969: The Effect of Tipped Domestic Refuse on Groundwater Quality: A Survey in North Kent. Water. Treat. Exam. 18: 18-34.

McLINN, E.L., PALMER, C.D., 1989: Laboratory Testing and Comparison of Specific-Conductance and Electrical-Resistivity Borehole - Dilution Devices. In: Proceedings of the Conference on New Field Techniques for Quantifying the Physical and Chemical Properties of Heterogeneous Aquifers, Dallas. F.J. Molz, J.G. Melville and O. Guven Eds., National Water Well Association, Water Well Journal Publishing Co., Ohio. 475 - 495.

MASCH, F.D., DENNY, K.J., 1966: Grain Size Distribution and its Effect on the Permeability of Unconsolidated Sands. Water Resources Research 2: 665-677.

MALCOCK, A.P., JOHNSON, R.W.M. (Eds.), 1971: The Feasibility of Compost Manufacture in Metropolitan Christchurch. Lincoln Papers in Resource

Management No.2, Lincoln College Press, Lincoln College, Canterbury, New Zealand. 40p.

MATHER, J.D., 1989: The Attenuation of the Organic Component of Landfill Leachate in the Unsaturated Zone: A Review. Quarterly Journal of Engineering Geology, London. 22: 241-246.

MATTHESS, G., 1972: Hydrogeologic Criteria for the Self Purification of Polluted Groundwaters. International Geological Congress 11: 296-302.

MATTHESS, G., 1982: The Properties of Groundwater. John Wiley & Sons, New York. 406p.

MAZAC, O., KELLY, W.E., LANDA, I., 1987: Surface Geoelectrics for Groundwater Pollution and Protection Studies. Journal of Hydrology 93: 277-294.

MAZAC, O., LANDA, I., KELLY, W.E., 1989: Surface Geoelectrics for the Study of Groundwater Pollution - Survey Design. Journal of Hydrology

MELVILLE, J.G., MOLZ, F.J., GUVEN, O., 1988: Field Experimental Methods in Stratified Aquifers. In: Ground-Water Contamination: Field Methods, ASTM STP 963, A.g. Collins and A.I. Johnson Eds., American Society for Testing and Materials, Philadelphia. 407-415.

MERKEL, R.H., 1972: The Use of Resistivity Techniques to Delineate Acid Mine Drainage in Ground Water. Ground Water 10: 38-42.

MERRICK, N.P., 1977: A Computer Program for the Inversion of Schlumberger Sounding Curves in Apparent Resistivity Domain. Hydrogeological Report No. 1977/5, Water Resources Commission of New South Wales, Australia.

MICHALSKI, A., 1989: Conductive Slug Tracing as a Single-Well Test Technique for Heterogeneous and Fractured Formations. In: Proceedings of the Conference on New Field Techniques for Quantifying the Physical and Chemical Properties of Heterogeneous Aquifers, Dallas, Texas. National Water Well Association, Water Well Journal Publishing Company. 247-263p.

MILDE, G., MOLLWEIDE, H.U., 1970: Hydrochemische Untersuchungen des Grubenwassers der Zechen der Huttenwerke Oberhausen AG. Berghau - Archiv 25(4): 21-27. Essen.

MOLZ, F.J., GUVEN, O., MELVILLE, J.G., 1983: An Examination of Scale-Dependent Dispersion Coefficients. Ground Water 21(6): 715-725.

MOLZ, F.J., MELVILLE, J.G., GUVEN, O., CROCKER, R.D., MATTESON, K.T., 1985: Design and Performance of Single-Well Tracer Tests at the Mobile Site. Water Resources Research 21(10): 1497-1502.

MOLZ, F.J., GUVEN, O., MELVILLE, J.G., 1989a: Characterization of the Hydrologic Properties of Aquifers: The Next Step. In: Proceedings of the Conference on New Field Techniques for Quantifying the Physical and Chemical Properties of Heterogeneous Aquifers, Dallas, Texas. F.J. Molz, J.G.

Melville and O. Guven Eds., National Water Well Association, Water Well Journal Publishing Co., Ohio. 407-418.

MOLZ, F.G., MORIN, R.H., HESS, A.E., MELVILLE, J.G., GUVEN, O., 1989b: The Impeller Meter for Measuring Aquifer Permeability Variations: Evaluation and Comparison with Other Tests. Water Resources Research 25(7): 1677-1683.

MOORE, C.R., 1990: Landuse Associated Groundwater Pollution in the Wairarapa Region. Unpublished M.Sc. Thesis, Victoria University of Wellington, New Zealand. 218p.

MORIN, R., 1986: Determining the Disturbance Effect of Well-casing Installation Methods in Unconsolidated Deposits using Geophysical Logs. EOS 67(44): 942.

MORIN, R.H., LeBLANC, D.R., TEASDALE, W.E., 1988: A Statistical Evaluation of Formation Disturbance Produced by Well-Casing Installation Methods. Ground Water 26(2): 207-217.

MOSS, R.L., 1988: Waste Survey for the Wellington Region. Wellington Regional Council Report, Wellington, New Zealand.

MULCOCK, A.P., JOHNSON, R.W.M. (Eds.), 1971: The Feasibility of Compost Manufacture in Metropolitan Christchurch. Lincoln Papers in Resource Management No.2 1971. Lincoln College Press, Lincoln College, Canterbury, New Zealand. 40p.

M.W.D., 1983: The Management and Disposal of Hazardous Wastes: A Discussion Document. Public Health Engineering Section, Civil Division, Ministry of Works and Development, Wellington. 54p.

MYERS, T.E., DUKE, D.C.Jr., MALONE, P.G., THOMPSON, D.W., 1979: Gas Production in Sanitary Landfill Simulators. In: Municipal Solid Waste Land Disposal. EPA-600-9-79-023a. United States Environmental Protection Agency, Cincinnati, Ohio. 58-73p.

NEW ZEALAND METEOROLOGICAL SERVICE 1983a: Summaries of Climatological Observations to 1980, New Zealand Meteorological Service Miscellaneous Publication 177, Ministry of Transport, Wellington, New Zealand. 102p.

NEW ZEALAND METEOROLOGICAL SERVICE 1983b: Rainfall Normals for New Zealand 1951 to 1980, New Zealand Meteorological Service Miscellaneous Publication 185, Ministry of Transport, Wellington, New Zealand. 102p.

NEW ZEALAND METEOROLOGICAL SERVICE, 1986: Summaries of Water Balance Data for New Zealand Stations. Ministry of Transport, Wellington, New Zealand. 102p.

NEWSOM, J.M., 1985: Transport of Organic Compounds Dissolved in Ground Water. Ground Water Monitoring Review Spring 28-36.

NICHOLSON, R.V., CHERRY, J.A., REARDON, E.J., 1983: Migration of Contaminants in Groundwater at a Landfill: A Case Study. 6. Hydrogeochemistry. Journal of Hydrology 63: 131-176.

NOONAN, M.J., CAMPBELL, A.S., SHAW, B., STANLEY, R.A., 1977: Investigation of Potential Pollution from a Proposed Landfill. Report to Christchurch Metropolitan Refuse Disposal Committee. Department of Agricultural Microbiology, Technical Publication No. 1, Lincoln College, New Zealand. 85p.

NORTH CANTERBURY CATCHMENT BOARD (N.C.C.B), 1983: Interim Report on the Groundwater Resources of the Central Plains. A Report Prepared by the Resource Investigation Division of the N.C.C.B. 124p.

PAPARUA COUNTY, 1982: Rehabilitation of Sand and Gravel Quarries in Paparua County. Stage 1 Investigation. 165p.

PARASNIS, D.S., 1979: Principles of Applied Geophysics, 3rd Edition. Chapman & Hall. 275p.

PAUL, D.G., PALMER, C.D., CHERKAUER, D.S., 1988: The Effect of Construction, Installation, and Development on the Turbidity of Water in Monitoring Wells in Fine-Grained Glacial Till. Ground Water Monitoring Review Winter 1988: 73-82.

PICKENS, J.F., GRISAK, G.E., 1981: Scale-Dependent Dispersion in a Stratified Granular Aquifer. Water Resources Research 17(4): 1191-1211.

POHLAND, F.G., 1980: Leachate Recycle as a Landfill Management Option. Journal American Society of Civil Engineering 106(EE6): 1057-1069.

POHLAND, F.G., DERIEN, J.T., GHOSH, S.B., 1983: 'Leachate and Gas Quality Changes During Landfill Stabilization of Municipal Refuse'. In: R.L. Wentworth (Ed.) Proceedings of the 4th International Symposium on Anaerobic Digestion, Boston, Dynatech, Cambridge, USA. 185-201p.

POHLAND, F.G., HARPER, S.R., 1986: Critical Review and Summary of Leachate and Gas Production for Landfills. United States Environmental Protection Agency, Cincinnati, Ohio. EPA/600/2-86-073.

POWELL, R.M., 1990: Total Organic Carbon Determinations in Natural and Contaminated Aquifer materials. In: Proceedings of the Fourth National Outdoor Action Conference on Aquifer Restoration, Ground Water Monitoring and Geophysical Methods, Las Vegas, Nevada. 1245-1258p.

PRICE, B., 1987: Cleaner Solutions from Rubbish Tips. New Scientist 25 June.

PRICE, M., 1985: Introducing Groundwater. Allen & Unwin Inc., Winchester Massachusetts. 195p.

PRATI, L. PAVANELLO, R., PESARIN, F., 1971: Assessment of Surface Water Quality by a Single Index of Pollution. Water Research 5: 741-751.

RAESIDE, J.D., RENNIE, W.F., 1974: Soils of Christchurch Region, New Zealand - The Soil Factor in Regional Planning. New Zealand Soil Survey Report 16, New Zealand Soil Bureau. 74p.

REFUTECH, 1986: Leachate Management in Sanitary Landfills. Refuse System Technology Consultants, Environmental Management Series.

REHFELDT, K.R., 1989: Application of the Borehole Flowmeter Method to Measure Spatial Variable Hydraulic Conductivity at the MADE Site. In: Proceedings of the Conference on New Field Techniques for Quantifying the Physical and Chemical Properties of Heterogeneous Aquifers, Dallas, Texas. F.J. Molz, J.G. Melville and O. Guven Eds., National Water Well Association, Water Well Journal Publishing Co., Ohio. 121-132.

RIGGS, C.O., HATHEWAY, A.W. 1988: Ground-Water Monitoring Field Practice - An Overview. In: Ground-Water Contamination: Field Methods, ASTM STP 963, A.g. Collins and A.I. Johnson Eds., American Society for Testing and Materials, Philadelphia. 121-136.

RISK, G.F., 1980: Tests of the Applicability of Electrical Prospecting Methods for Detecting Underground Seepage of Pulp and Paper Mill Effluent at Kinleith. Report No. 170, Geophysics Division, DSIR, Wellington, New Zealand.

ROBERTS, P.V., GOCTZ, M.N., MACKAY, D.M., 1986: Natural Gradient Experiment on Solute Transport in a Sand Aquifer. 3. Retardation Estimates and Mass Balances for Organic Solutes. Water Resources Research 22(13): 2047-2058.

ROBINSON, W.D., 1986: The Solid Waste Handbook: A Practical Guide. John Wiley & Sons, New York. 811p.

RODRIGUES, E.B., 1987: Application of Gravity and Seismic Methods in Hydrogeological Mapping at a Landfill Site in Ontario. In: Aquifer Restoration, Ground Water Monitoring and Geophysical Methods - First National Outdoors Action Conference, Association of Ground Water Scientist, National Water Well Association. 487-504p.

ROGERS, R.B., KEAN, W.F., 1980: Monitoring Ground-Water at a Fly Ash Disposal Site using Surface Electrical Resistivity Methods. Ground Water 18(5): 472-478.

ROGERS, D.A., TOTTEN, R.R., 1988: The Composition of Leachates from Several New Zealand Landfills. NECAL Report Series 88/5. National Environmental Chemistry and Acoustics Laboratory, Department of Health, Auckland, New Zealand. 35p.

ROUSE, B.L., 1977: Production of Leachate from Deposits of Solid Waste and its Attenuation through the Substrata. Unpublished M.Sc. Thesis, Lincoln College, New Zealand.

ROY, A., APPARAO, A., 1971: Depth of Investigation in Direct Current Methods. Geophysics 36(5): 943-959.

ROY, A., 1972: Depth of Investigation in Wenner, Three-Electrode and Dipole-Dipole DC Resistivity Methods. Geophysical Prospecting 20: 329-340p.

RUMBAUGH, J.O., CALDWELL, J.A., SHAW, S.J., 1987: A Geophysical Ground Water Monitoring Program for a Sanitary Landfill: Implementationand Preliminary Analysis. In: Aquifer Restoration, Ground Water Monitoring and Geophysical Methods, First National Outdoors Action Conference. Association of Ground Water Scientists, National Water Well Association. 623-641p.

RUSSELL, G.M., HIGER, A.L., 1988: Assessment of Ground-Water Contamination near Lantana Landfill, South Florida. Ground Water 26(2): 156-164.

SAINES, M., 1981: Errors in Interpretation of Ground-Water Level Data. Ground Water Monitoring Review Spring 56-61.

SALTZMAN, S., YARON, B., 1986: Pesticides in Soil. Van Nostrand Reinhold Soil Science Series, Van Nostrand Reinhold Co. Inc. 379p.

SCHARCH, J.F., 1985: Water Balance Analysis Program for the IBM PC Micro-computer. Bureau of Solid Waste Management, Wisconsin Department of Water Resources, Madison, Wisconsin. 67p.

SCHOEDER, P.R., MORGAN, J.M., WALSKI, T.M., GIBSON, A.C., 1984: The Hydrologic Evaluation of Landfill Performance (HELP) Model Vol. 1 Users Guide for Version 1. EPA/530-SW-84-009, Environmental Protection Agency, Washington DC. 120p.

SCHOFIELD, J.C., 1960: Boron in Some New Zealand Ground Waters. New Zealand Journal of Geology and Geophysics 3: 98-104.

SCHWARTZ, F.W., 1988: Contaminant Hydrogeology - Dollars and Sense. Journal of Hydrology 100: 453-470.

SCOTT KEYS, W., MacCARY, L.M., 1971: Application of Borehole Geophysics to Water-Resources Investigations. Techniques in Water-Resources Investigations of the United States Geological Survey, Book 2, Chapter E1. U.S. Department of the Interior, U.S. Government Printing Office, Washington, 126p.

SHACKELFORD, W.M., KEITH, L.H., 1976: Frequency of Organic Compounds Identified in Water. Analytical Chemistry Branch, Environmental Research Laboratory, Athens, Ga.

SHARMA, P.V., 1976: Geophysical methods in Geology. In: Methods in Geochemistry and Geophysics No. 12: 261-293.

SHEPHERD, R.G., 1989: Correlations of Permeability and Grain Size. Ground Water 27(5): 633-638.

SHERRIFF, J.M., 1988: Canterbury Regional Council Waste Management Investigation 1987/88, Report One, Part One 'Waste Stream Survey's, and Part Two 'Assessment of Existing Disposal Systems', Christchurch Metropolitan Refuse Disposal Committee.

SIEGEL, D.I., STONER,D., BYRNES,T., 1990: A Geochemical Process Approach to Identify Inorganic and Organic Groundwater Contamination. In: Proceedings of the Fourth National Outdoors Action Conference on Aquifer Restoration, Ground Water Monitoring and Geophysical Methods, Las Vegas, Nevada. National Water Well Association, Water Well Journal Publishing Company. 1291-1301.

SLICHTER, C.S., 1902: The Motion of Underground Waters. US Geol. Survey Water Supply Paper No. 67. 106p.

STAPLEDON, D.H., 1979: Investigation and Characterization. Extension Course on 'Tunnelling Design and Practice' Australian Geomechanics Society, Melbourne, October 1979. 13-33pp.

STEWART, M., BRETNALL, R., 1986: Interpretation of VLF Resistivity Data for Ground Water Contamination Surveys. Ground Water Monitoring Review 6(1): 71-75.

STIERMAN, D.J., 1984: Electrical Methods of Detecting Contaminated Groundwater at the Stringfellow Waste Disposal Site, Riverside County, California. Environmental Geology Water Science 6(1): 11-20.

STIERMAN, D.J., RUEDISILI, L.C., 1988: Integrating Geophysical and Hydrogeological Data: An Efficient Approach to Remedial Investigations of Contaminated Ground Water. In: Ground Water Contamination: Field Methods, ASTM STP 963, A.G. Collins and A.I. Johnson Eds., American Society for Testing and Materials, Philadelphia. 43-57p.

STOLLAR, R.L., ROUX, P., 1975: Earth Resistivity Surveys - a Method for Defining Ground-Water Contamination. Ground Water 13(2): 145-150.

STONE, L., 1990: Toxics Campaign. In: Action Against Water Pollution in Canterbury, Healthy Cities Conference in Association with the Christchurch School of Medicine. R.A. Allardyce (Ed.) 28-31p.

SUDICKY, E.A., CHERRY, J.A., FRIND, E.O., 1983: Migration of Contaminants in Groundwater at a Landfill: A Case Study. 4. A Natural-Gradient Dispersion Test. Journal of Hydrology 63: 81-108.

SUDICKY, E.A., 1986: A Natural Gradient Experiment on Solute Transport in a Sand Aquifer: Spatial Variability of Hydraulic Conductivity and its Role in the Dispersion Process. Water Resources Research 22(13): 2069-2082.

SUGGATE, R.P., 1958: Late Quaternary Deposits of the Christchurch Metropolitan Area. New Zealand Journal of Geology and Geophysics 1: 103-122.

SUGGATE, R.P., 1963: The Fan Surfaces of the Central Canterbury Plains, New Zealand. New Zealand Journal of Geology and Geophysics 6: 281 -287.

SUGGATE, R.P., 1965: Late Pleistocene Geology of the Northern Part of the South Island, New Zealand. New Zealand Geological Survey Bulletin 77.

SUGGATE, R.P., 1973: Sheet 21 - Christchurch. 2nd Edition. Geological Map of New Zealand 1:250,000, Wellington, Department of Scientific and Industrial Research.

SWARTZ, J.H., 1937: Resistivity Studies of some Salt-water Boundaries in the Hawaiian Islands. Transactions of the American Geophysical Union 18: 387-393.

SWEENEY, J.J., 1984: Comparison of Electrical Resistivity Methods of Investigation of Ground Water Conditions at a Landfill Site. Ground Water Monitoring Review 4(1): 52-59.

SYNERGY APPLIED RESEARCH LTD., 1985: Estimated New Zealand Waste Quantities: Glass, Paper, Ferrous Metal and Plastic. Resource Conservation Series 19. Department of Trade and Industry, Wellington, New Zealand. 47p.

TALBOT, J., WEEBER, J.H., FREEMAN, M.C., MASON, C.R., WILSON, D.D., 1986: The Christchurch Artesian Aquifers. A report prepared by the Resources Division of the N.C.C.B. and R.W.B., Christchurch, New Zealand. 160p.

TAYLOR, C.B., WILSON, D.D., BROWN, L.J., STEWART, M.K., BURDEN, R.J., BRAILSFORD, G.W., 1989: Sources and Flow of North Canterbury Plains Groundwater, New Zealand. Journal of Hydrology 106: 311-340.

TAYLOR, G., 1991: Waste Management: Chaos or Coordination? New Zealand Engineering, March.

TAYLOR, K., 1989: Review of Borehole Methods for Characterizing the Heterogeneity of Aquifer Hydraulic Properties. In: Proceedings of the Conference on New Field Techniques for Quantifying the Physical and Chemical Properties of Heterogeneous Aquifers, Dallas, Texas. F.J. Molz, J.G. Melville and O. Guven Eds., National Water Well Association, Water Well Journal Publishing Co., Ohio. 121-132.

TAYLOR, K., HESS, J., MAZZELLA, A., HAYWORTH, J., 1990a: Comparisons of Three Methods to Determine the Vertical Stratification of Pore Fluids. Ground Water Monitoring Review Winter, 91- 95.

TAYLOR, K., WHEATCRAFT, S., HESS, J., HAYWORTH, J., MOLZ, F., 1990b: Evaluation of Methods for Determining the Vertical Distribution of Hydraulic Conductivity. Ground Water 28(1): 88-98.

TAYLOR, S.R., MOLTYANER, G.L., HOWARD, K.W.F., KILLEY, R.W.D., 1987: A Comparison of Field and Laboratory Methods for Determining Contaminant Flow Parameters. Ground Water 25(3): 321-330.

TELFORD, W..M., GELDART, L.P., SHERIFF, R.E., KEYS, D.A., 1976: Applied Geophysics. Cambridge University Press, London. 860p.

THOMPSON, C.S., 1985: New Zealand Meteorological Service Miscellaneous Publication 187. Ministry of Transport, Wellington, New Zealand. 58p.

THOMPSON, F.L., TAYLER, S.W., 1984: Comparison of Two Groundwater Flow Models - UNSAT1D and HELP. Prepared by Battelle, North West Laboratories, for Electric Power Research Institute, Pal Alto, California.

*

THORNTHWAITE, C.W., MATHER, J.R., 1957: Instructions and Tables for Computing Potential Evapotranspiration and the Water Balance. Publication in Climatology, Laboratory of Climatology, Drexel Institution of Technology, Centerton, New Jersey. 10(3): 185-311.

THORPE, H.R., BURDEN, R., COULING, K., HAWKINS, N., O'DONNELL, S., RICHARDS, G., SCOTT, D., 1977: Movement of Contaminants into and through the Heretaunga Plains Aquifer. NWASCO, Ministry of Works and Development Water and Soil Division, Napier, New Zealand. 148p.

THORPE, H.R., BURDEN, R.J., SCOTT, D.M., 1982: Potential for Contamination of the Heretaunga Plains Aquifers. Water and Soil Technical Publication No. 24. 149p.

THORPE, H.R., SCOTT, D.M., 1991: Groundwater of the Canterbury Plains, New Zealand. In: Proceedings of the ASCE 'Groundwater in the Pacific Rim' Symposium, Hawaii.

THORSTENSEN, L., 1990: Solid Waste Management in New Zealand; Current Trends and Future Directions. In: Waste Management Institute New Zealand Inc. Second Annual Conference, Palmerston North. 257-271p.

TODD, D.K., 1980: Groundwater Hydrology, 2nd Edition. John Wiley & Sons Inc., New York. 535p.

UMA, K.O., EGBOKA, B.C.E., ONOUHA, K.M., 1989: New Statistical Grain-size Method for Evaluating the Hydraulic Conductivity of Sandy Aquifers. Journal of Hydrology 108: 343-366.

URISH, D.W., 1983: The Practical Application of Surface Electrical Resistivity to Detection of Ground-Water Pollution. Ground Water 21(2): 144-152.

URISH, D.W., 1985: Electrical Resistivity Study at Testhole Locations 76-T, 76-U and 76-32 in the lower Wood River Basin, Rhode Island. Unpublished Report of the Rhode Island Water Resource Board.

U.S.E.P.A., 1973: An Environmental Assessment of Potential Gas and Leachate Problems at Land Disposal Sites. 33p.

UNITED NATIONS ENVIRONMENTAL PROGRAMME (U.N.E.P.), 1988: Hazardous Waste Management. In: Industry and Environment 11(1).

VAN DAM, J.C., MEULENKAMP, J.J., 1967: Some Results of the Geoelectrical Resistivity Method in Ground Water Investigations in The Netherlands. Geophysical Prospecting 15: 92-115.

VAN OVERMEEREN, R.A., RITSEMA, I.L., 1988: Continuous Vertical Electrical Sounding. First Break 6(10): 313-324.

WARREN, D., 1989: Environmental Impacts of Waste. In: Waste Management Institute New Zealand Inc. Conference, Nov. 1, Wellington, New Zealand.

WEIST, R., 1979: Handbook of Chemistry and Physics, 60th Edition. C.R.C. Press Inc.

WHITE, P.A., 1985: Shallow Groundwater Flow Direction and Velocity at the Takaka Sewage Pond, Nelson. Water and Soil Report No. WS1014, Ministry of Works and Development, Christchurch, New Zealand. 10p.

WHITE, P.A., 1986: Shallow Ground Water Flow Direction and Velocity at the Renwick Refuse Tip, Marlborough, Feb. 1986. Water and Soil Report No. WS 1109, Ministry of Works and Development, Christchurch, New Zealand. 12p.

WHITE, P.A., 1988: Measurement of Ground Water Parameters using Salt Water Injection and Surface Resistivity. Ground Water 26(2): 179-186.

WHITE, P.A., 1991: Experiments for Ground Water Flow Direction and Velocity Measurements with Salt Water Injection and Surface Resistivity. D.S.I.R. Internal Report.

WHITTLE, J.H., SWANSON, A.C.S., 1986: Site Investigation Techniques used to Assess the Likely Hydrological Impact of a Proposed Landfill Site. In: Groundwater in Engineering Geology, London Geological Society Engineering Geology Special Publication No. 3, 197-205p.

WILSON, D.D., 1985: Erosional and Depositional Trends in Rivers of the Canterbury Plains, New Zealand. New Zealand Journal of Hydrology 24: 32-44.

WOESSNER, W.W., 1988: Drill-Through Casing Driver Drilling Method for Construction of Monitoring Wells in Coarse, Unconsolidated Sediments. Ground Water Monitoring Review Winter 1988: 69-72.

WORLD RESOURCES INSTITUTE, 1990: World Resources 1990-91; A Guide to the Global Environment. Oxford University Press.

XERTEX, 1986: DC-90 Combustion Infrared Laboratory Total Organic Carbon Analyser. Dohrman Equipment Manual, 3rd Edition.

YOUNG, P.J., BALDWIN, G., WILSON, D.C., 1984: Attenuation of Heavy Metals Within Municipal Waste Landfill Sites. In: Hazardous and Industrial Waste Management and Testing: Third Symposium. ASTM STP 851, L.P. Jackson, A.R. Rohlik and R.A. Conway, Eds., American Society for Testing and Materials, Philadelphia, 1984. 192-212p.

YOUNG, S.C., WALDROP, W.R., 1989: An Electromagnetic Borehole Flowmeter for Measuring Hydraulic Conductivity Variability. In: Proceedings of the Conference on New Field Techniques for Quantifying the Physical and Chemical Properties of Heterogeneous Aquifers, Dallas, Texas. F.J. Molz, J.G. Melville and O. Guven Eds., National Water Well Association, Water Well Journal Publishing Co., Ohio. 463-474.

ZANONI, A.E., 1971: Groundwater Pollution and Sanitary Landfills - a Critical

Review. Proceedings of the National Groundwater Quality Symposium. 97-107p.

ZANONI, A.E., 1973: Potential for Groundwater Pollution from Land Disposal of Solid Wastes. Critical Reviews in Environmental Control 3(3): 225p.

ZOHDY, A.A.R., EATON, G.P., MABEY, D.R., 1974: Application of Surface Geophysics to Ground-Water Investigations. Chapter D1, Book 2, U.S. Geol. Survey Techniques of Water Resource Investigations. 116p.